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Nekton Utilization of Intertidal Fringing Salt Marsh and Revetment Hardened Shorelines

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NEKTON UTILIZATION OF INTERTIDAL FRINGING SALT MARSH AND REVETMENT HARDENED SHORELINES

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DEDICATION

This thesis is dedicated to all the mysterious critters that burble, bubble, snap, drip, spit, flip, swish, splash, whack, dribble, puddle, paddle and hum in Sarah Creek on dark, still, hot summer nights. Nobody really knows, as Dr. Seuss said, what goes on down below.

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ABSTRACT

Flume nets were used to quantitatively sample nekton from narrow intertidal fringe marsh (3 meters wide) and intertidal riprap revetment shoreline in Sarah Creek, Virginia. Physical site parameters were similar in both habitats except for intertidal area flooded, which was approximately 58% larger in the fringe marsh due to the greater slope that is commonly associated with riprap revetment structures. A total of 2,233 fish from 19 species, 1,150 *Callinectes sapidus* and 30,768 *Palaemonetes pugio* were captured during the study. Sixteen species of fishes were captured in the fringe marsh, and 11 were captured in the riprap. Significantly greater abundance of *Fundulus heteroclitus*, *Menidia menidia*, *Paralichthyes dentatus*, *Fundulus majalis*, *Morone americana*, *Leiostomus xanthurus*, total fish and *Callinectes sapidus* were captured per meter of intertidal fringe marsh shoreline. *P. dentatus*, *M. americana*, and *L. xanthurus* and *F. majalis* were captured almost exclusively in the fringe marsh. Commercial and recreationally important fish abundance averaged 1.8 individuals per linear meter of fringe marsh edge (SE = 0.59) and 0.2 individuals per linear meter of riprap edge (SE = 0.05). *Gobiosoma bosc* and *P. pugio* used both habitats to a similar degree. Significantly larger *F. heteroclitus*, *C. sapidus* and *G. bosc* were capture in the riprap. Thus, biomass per meter of shoreline for *F. heteroclitus* and *C. sapidus,* both dominant nekton species in this study, were similar between habitats, and biomass per meter of shoreline of *G. bosc* was greater in riprap. Biomass per meter of shoreline was significantly greater in the fringe marsh for *M. menidia*, *P. dentatus*, *F. majalis*, *L. xanthurus* and total fish. Fish abundance and biomass was more evenly distributed among fish species in the fringe marsh. Riprap did not attract any species, in significant numbers, that were not also found in the fringe marsh. While dominant estuarine resident species were captured in riprap, results show that fringing marsh is a more highly utilized intertidal habitat. This is especially true for commercially and recreationally important fishes and blue crabs. High abundance and density of nekton in the intertidal fringe marsh compared to other studies highlights the importance of fringe marsh habitat and tidal estuarine creeks to the nekton community.

NEKTON UTILIZATION OF INTERTIDAL FRINGING SALT MARSH AND REVETMENT HARDENED SHORELINES

INTRODUCTION

Shoreline Hardening

Stone revetment structures, or riprap, consist of a graded slope of large stone, often in two layers, used to anchor the shoreline foot in order to slow or stop shoreline erosion (Hardaway and Byrne 1999). The shoreline protection function derives from the armoring characteristic of the rock and the rocks' ability to dissipate wave energy (Mulvihill et al. 1980). Hardaway and Byrne (1999) refer to revetment as a preferred stabilization technique to prevent shoreline erosion along low wave energy shorelines with eroding upland banks. Properly designed revetment can function for over 50 years. Revetment use increased in the 1970's and has gained in popularity over bulkheads in the last twenty years, especially in rural areas (Priest et al. 1990, Hardaway and Byrne 1999).

The incremental hardening of natural shorelines with rock, a habitat not naturally found in the Chesapeake Bay and its tidal tributaries, comprise a significant component of the shoreline in some waterways over time (Roman and Nordstrom 1996). Close to 483 kilometers of Chesapeake Bay shoreline has been armored in the last 20 years (Titus 1998). Virginia has 16,073 kilometers of tidal shoreline, not including the 212 kilometers of open ocean shorefront (www.vims.edu/ccrm/gis/gisdata.html). An average of 20.4 kilometers of shoreline were hardened with riprap and 7.7 kilometers of shoreline were hardened with bulkhead every year between 1993 to 2000 (Barnard et al. 2001). A cumulative total of 195 kilometers of Virginia shoreline have been hardened over that eight-year time period (Barnard et al. 2001). Permits were granted for the construction of 6.18 km of shoreline structures in Gloucester County between 1989 and 1994. Riprap comprised 53% of this total (Berman et al. 1998). Almost 2.2 kilometers of shoreline were hardened along the York River in 1999 alone (State of the York Watershed 2000). In 2000, Virginia received over 1,100 applications for wetland activity permits (Barnard et al. 2001) and permits were issued for the construction of 26.3 linear kilometers of rock riprap (www.vims.edu/rmap/wetlands/cgi-bin/staff/detail.htm). Forty-percent (21.1 km / 52.8 km) of the shoreline surrounding Gloucester Point, including Sarah Creek have been hardened (Berman et al. 1999).

The majority of permitted shoreline stabilization projects impact non-vegetated intertidal areas. Those that do impact vegetated areas, such as saltmarsh cordgrass and brackish water mixed vegetation communities, usually result in loss of the vegetation even though those marshes are protected under the Virginia Wetland Act of 1972 (Priest et al. 1990, Barnard et al. 2001). Between 1989 and 1994, 4.89 acres of tidal wetlands were lost in Gloucester County in areas where permitted shoreline structures were installed. Sixty-seven percent of this loss (8,103 square feet) occurred in areas where riprap was installed, the remainder occurred during permitted bulkhead projects (Berman et al. 1998). Fringe marsh vegetation is often the type of wetland habitat lost during hardening due to its wide scale occurrence in the Chesapeake Bay region (Priest et al. 1990, Berman et al.). Construction of revetment can destroy narrow fringing marshes by covering them up, and larger marsh areas may be lost due to altered water circulation (Mulvihill et al. 1980). Loss of tidal marsh vegetation and its associated values is the most significant ecological impact of riprap construction (Mulvihill et al.1980).

The detrital food chain and vital edge habitat is reduced when fringing marsh is lost, potentially impacting estuarine-dependent fishery resources (Herke et al. 1992). Shallow water edge areas are a refuge for juvenile blue crabs and small fish and are a critical habitat for all life stages of killifish and grass shrimp in the Chesapeake Bay (Dittel et al. 1995). Altering or restricting access to saltmarsh habitats can be detrimental to the production and maintenance of resident nekton by disrupting the feeding patterns and movements of estuarine fish that have adapted to the daily tidal flow into the intertidal shallows and marsh edge (Peterson and Turner 1994, Hendon et al. 2000). This could lead to declines in the amount of production that moves from shallow to deeper waters through a chain of predator-prey interactions described by the trophic relay concept (Kneib 1997). Prey consumption and trophic export has been found to be greater along both depositional and erosional marsh edges than interior marshes or the unvegetated intertidal (Cicchetti and Diaz 2000). Growth rates of mummichogs, a key shallow water estuarine trophic relay species, can decease below detection limits when they are denied access to intertidal marsh (Weisburg and Lotrich 1982). Marsh vegetation values, including essential habitat for migratory birds and water filtering of upland runoff and groundwater are also lost when fringing marsh vegetation is destroyed (Wohlgemuth 1990, Roman and Nordstrom 1996).

Shoreline hardening to protect property can be expected to continue in light of an accelerating pace of shoreline development and shoreline retreat (Stevenson and Kearney 1996, Titus 1998). Development has increased by 17% along the Chesapeake Bay's shores in the past several decades. The average horizontal shoreline erosion rate along the York River, Virginia is 0.12 m per year on the North bank and 0.36 m per year on the South bank (Hardaway and Byrne 1999). Twenty percent of the Bay shoreline is retreating at a rate of 2.0 m per year and most areas lose at least 0.5 m of shore every year (Stevenson and Kearney 1996).

The increasing rates of shoreline erosion in the Chesapeake Bay mirror a general trend in sea level rise. Sea level rise in the Chesapeake Bay is among the highest along the Atlantic Coast, with a rate of 3.6 mm per year near the Bay's mouth (Stevenson and Kearney 1996), or at least 0.3 m per century (Hardaway and Byrne 1999). The trend in sea level rise began accelerating in 1850, with the largest increases seen after 1920 (Hardaway and Byrne 1999).

Intertidal marshes grow vertically and laterally landward as sea level rises (Orson et al. 1985, Hardaway and Byrne 1999). Some marshes in the Chesapeake Bay do not have a large enough vertical accretion rate to keep up with present rates of sea level rise (Stevenson and Kearney 1996). These marshes are further destabilized when they are cut off from upland sand supply by shoreline protection structures placed landward of existing marsh. Shoreline stabilization projects may also take away the sand supply that creates beaches, spits and offshore bars when uplands naturally erode (Hardaway and Byrne 1999). Human-made structures placed along the high-tide mark prevent landward marsh migration and re-creation of estuarine environments as sea level rises. Existing marsh becomes constrained between the structure and the rising water in what is termed the 'coastal squeeze' (Nordstrom and Roman 1996).

Riprap shoreline hardening has also been used in freshwater settings. Relatively few studies have looked into the impact that riprap has on fresh water nekton community, and results have been mixed. In one study, species diversity was found to be greater along riprap shorelines compared to bulkhead and natural banks in fresh water lakes in Wisconsin (Jennings et al. 1999). The authors attributed this difference to greater habitat complexity due to interstices in the rock riprap. Similarly, abundance of several fish species was found to be greater in shorelines with rock crib structures in Lake Tahoe (Beauchamp et al. 1994). Revetment hardened shorelines in riverine systems do provide habitat for juvenile salmonids. However, the occurrence of undercut banks, low overhead cover and woody debris, all critical habitats to older salmonid life stages, is lower along hardened river banks compared to natural vegetated banks (Schmetterling et al. 2001). Riprap is therefore believed to have an overall negative effect on salmonid habitats in riverine systems, and makes riverine restoration more difficult.

The effects of shoreline hardening on shallow water estuarine nekton communities has received even less attention in the literature, possibly due to sampling difficulty in the intertidal zone. The abundance of estuarine nekton was consistently lower in shallow waters adjacent to bulkheaded shorelines compared to natural shorelines in a New Jersey lagoon (Byrne 1995). Investigations regarding nekton use of large, subtidal marine jetties indicate that the complexity of these structures attracts inshore oyster reef and offshore hard bottom nekton, acting as small-scale nurseries (Hay and Sutherland 1988). The reef-like attributes of riprap structures may be questioned in intertidal areas, as the habitat complexity provided by anthropogenic structure is not always a positive attribute to aquatic systems. Peterson et al. (2000) found significantly less nekton abundance and diversity in waters directly adjacent to marsh that had been altered with bulkheading or stone riprap compared to unaltered marsh edges in Mississippi embayments. Similarly, Able et al. (1998) found decreased nekton abundance and diversity in subtidal areas under large pier structures in the lower Hudson River.

Marsh Utilization

The marsh edge ecotone is defined as the transitional zone from shallow open water in subtidal habitats through periodically flooded emergent vegetation to high intertidal salt marsh habitats (Rakocinski et al. 1992). Intertidal fringing marsh edge is a conspicuous feature of the marsh edge ecotone. Research continues to highlight the importance of the marsh edge ecotone in estuarine systems as a nursery, refuge and foraging ground for estuarine resident and transient nekton (Kneib 1982, Kneib and Stiven 1982, Zimmerman and Minello 1984, McIvor and Odum 1988, Rakocinski et al. 1992, Baltz et al. 1993, McIvor and Rozas 1996, Cicchetti 1998a, Cicchetti and Diaz 2000, Halpin 2000). Two other primary estuarine nursery areas are submerged aquatic vegetation beds (SAV) and low salinity areas near the estuary headwaters (Deegan 1989). Many studies have focused solely on nekton use of the flooded intertidal salt marsh vegetation portion of this ecotone, highlighting heavy and dynamic use of salt marsh habitat (McIvor and Odum 1986, Hettler 1989, Rozas and Reed 1993, Minello et al. 1994, Peterson and Turner 1994, Yozzo et al. 1994, Ayers 1995, Minello and Webb 1997).

Distribution of nekton in the salt marsh is related to a host of interrelating biotic and abiotic factors such as forage base and refuge value (Boesch and Turner 1984, Ruiz et al. 1993), tidal inundation patterns (Rozas 1995), and temperature, salinity, depth, substrate, dissolved oxygen, currents and turbidity (Rakocinski et al. 1992, Baltz et al. 1993). In general, greater abundances of nekton are found closer to the marsh edge (Kneib and Wagner 1994, Peterson and Turner 1994, Cicchetti 1998a); in lower order marsh streams - especially marsh rivulets (Rozas and Odum 1987, McIvor and Odum

1988, Rozas et al. 1988, Hettler 1989); in more reticulated marshes with greater edge habitat (Minello et al 1994); and along marshes with a depositional, or gradually sloping edge (McIvor and Odum 1988).

The marshes sampled in this study are very narrow compared to the much larger marshes sampled in the majority of published work. Rozas (1992) found narrow fringing marshes between 5 – 8 meters wide to be equally important high-tide habitat for nekton as wider (> 20 m.) marshes in Louisiana. Cicchetti (1998c) found a large abundance of nekton, including spot and blue crabs, along erosional marsh edge at the Goodwin Islands, Virginia. Nekton biomass in depositional marsh edge at the Goodwin Islands, York River, Virginia was statistically similar to that in SAV habitats at high tide (Cicchetti 1998b). As SAV beds in the Chesapeake Bay have declined, shallow areas, and bordering *Spartina* marshes may play an increasingly important role as a juvenile fish and blue crab refuge (Ruiz et al. 1993, Peterson and Turner 1994). These findings suggest that narrow fringing marsh, a habitat overlooked in marsh utilization studies, may be of significant importance to the shallow water nekton community. Fringe marsh sampling in this study fills the need for further quantitative salt marsh utilization studies as suggested by Rozas (1995), McIvor and Rozas (1996) and Minton (1999).

Mulvihill et al. (1980) and Nordstrom and Roman (1996) identify the need to investigate the effect of shoreline protection structures on the biological community, especially in areas where this type of habitat did not formerly exist. Research comparing the habitat value of fringing marsh to intertidal riprap will give managers information that can be used to make informed decisions concerning future shoreline hardening permit processing and policy. The goal of this project is to quantify and compare nekton utilization of intertidal fringing salt marsh (2 - 4 meters wide) and intertidal revetment shorelines in Sarah Creek, Virginia. The objectives are: (1) to quantify the value of narrow fringing marsh habitat to shallow water nekton; (2) to quantify the value of riprap habitat to shallow water nekton; and (3) to quantify the effects of replacing fringing marsh habitats with riprap habitats on estuarine nekton.

METHODS

Site Description

This study was performed in Sarah Creek, a tributary to the lower York River near Gloucester Point, Virginia (37°15'30" N, 76°29'00" W), approximately 10 km upriver from the open Chesapeake Bay and 40 km from the Atlantic Ocean (Figure 1). Sarah Creek was chosen for the study due to the presence of substantial shoreline hardening, and its proximity to the Goodwin Islands component of the Chesapeake Bay National Estuary Research Reserve in Virginia (CBNERRVA) and the Virginia Institute of Marine Science. The Goodwin Island NERR site is a relatively undisturbed area approximately 5 nautical miles from Sarah Creek. A number of nekton utilization studies have been conducted on the Goodwin Islands. The majority of the Sarah Creek shoreline is comprised of residential development. Average fetch lengths in Sarah Creek do not exceed one nautical mile, categorizing the shorelines as low-energy (Hardaway and Byrne 1999). Average tidal amplitude in Sarah Creek is 0.75 m (Varnell and Havens 1995).

Sampling was conducted with flume nets at three paired sites, each with intertidal fringing salt marsh and revetment (Figure 2). Paired sites were chosen based on similarity of offshore slope, shoreline exposure, length of shoreline type and proximity of paired fringe marsh and riprap. Narrow fringing marshes less than 4 meters wide were chosen for marsh sample sites in order to make appropriate comparisons between fringe marsh and revetment hardened shoreline. Landowner permission was obtained for the eight properties used during the study.

The fringing marsh at sites A and B were comprised of *Spartina alterniflora* in the lower areas and mixed mesohaline wetland plants, including *Spartina patens*, further from the marsh edge (Table 1). The marsh at site C was comprised totally of *Spartina alterniflora*. All revetments were predominately constructed of class A1 rock (25 – 75 pounds, with no more than 10% of the stones weighing more than 75 pounds) and had been in place for more than five years (VMRC 1999). Marsh periwinkles (*Littorina irrorata*) and scattered oysters (*Crassostrea virginica*) were present at all study sites. Numerous ribbed mussels (*Geukensia demissa*) were found growing in the interstices of the revetment rock. Neither submerged aquatic vegetation nor extensive macroalgal growth was found near any of the study sites. The owner of the fringing marsh at site C has a permit in hand to install 61 meters of revetment to protect the eroding bank located behind the sampled fringing marsh. The eroding bank behind the fringing marsh at site A makes this marsh a candidate for future shoreline stabilization as well.

Marsh width was determined at each site as the mean width of ten randomly selected transects running perpendicular to the shoreline. *Spartina alterniflora* stem density was based on replicate (N = 10) haphazardly placed 0.25 m^2 quadrats at each fringe marsh site. Surficial (upper 2 cm) sediment texture was determined from three composited core samples taken from the fringe marsh or riprap edge at each site. Grain size mass ratios for surficial sediment (upper 2 cm) were determined by wet sieve and pipette analysis, and results were expressed on the Wentworth grain scale (Folk 1980). The Wentworth grain scale was used to separate gravel, sand, silt and clay size classes.

Differential surveying techniques were used to determine offshore slope, as defined from the creekside edge of the fringe marsh or riprap to 15m offshore. Survey measurements were taken from the approximate spring high tide level to the creekside edge of the marsh or riprap to calculate the slope of the fringe marsh or surface of the riprap. The slope of the ground beneath the regularly flooded riprap was calculated when the rocks were moved during flume net construction.

Relative tidal height at the creekside edge of the flume net and relative flood distance from the flume net edge to dry land at each net station was recorded immediately after block net deployment. Distance flooded into the revetment was calculated in advance for all tidal heights when rocks were moved away during construction using a vertical meter stick, a level line and by solving for the hypotenuse of resulting field measurements at each riprap flume net station. These measurements were not corrected between flume nets or to mean low water or any other baseline.

An Endeco YSI Environmental Monitoring System PC6000 datasonde was used to record temperature (°C), salinity, turbidity (NTU) and dissolved oxygen ($mg \cdot L^{-1}$). The datasonde was hung at the edge of the sampled intertidal habitats at slack before ebb tide. The datasonde was deployed for 15 minutes in both the fringing marsh and the riprap intertidal area during the majority of sample tides. Site parameters are listed in Table 2 and 3.

Flume Net Design

The flume net, as initially described by McIvor and Odum (1986), was chosen for sampling gear after evaluating possible collection methods. The flume net was the only collection gear that could directly quantify nekton use of revetment. Modified flume nets are often used to collect nekton from intertidal marshes (McIvor and Odum 1986, Hettler

1989, Peterson and Turner 1994). Rozas and Minello (1997) highly recommend the flume net for tidal marsh sampling in a summary review of shallow estuarine habitat collection methods. McIvor and Odum (1986) also recommend the flume net for comparing different tidal marsh habitats.

Flume nets sample a known area and have high and easily measured nekton recovery efficiencies. They can only be used, however, where sampled habitats drain completely and no standing water remains at low tide (Rozas and Minello 1997). The permanent sides of the net do create structure, and these sides also restrict access to the sampling area to one direction. However, it is generally assumed in highly structured habitats that the additional structure neither attracts nor deters normal nekton use, and that any bias is equal between the two habitats studied (McIvor and Odum 1986, Peterson and Turner 1994). I assumed here that the specific stations chosen for flume nets are representative of the marsh or riprap in Sarah Creek. McIvor and Odum (1986) report that flume nets catch representative samples of the intertidal nekton species assemblage, and that flume nets sample in the most unbiased manner possible given the challenges associated with sampling in a marsh.

Flume nets (Figure 3) consisted of two parallel 0.9 m high sides, or 'wings,' made with marine grade filter cloth material. These permanent wings were oriented perpendicular to the shoreline and were stabilized with 1.8 m long 5cm x 5cm wood stakes. Flume net sides were left in place for the duration of the sampling period, consistent with the methods of other flume net studies (McIvor and Odum 1986, Peterson and Turner 1994, Able and Hagan 2000). Steel staples (15 cm x 2.5 cm) hammered every 5 cm secured a flap of the filter cloth wall to the ground, sealing the flume net wings firmly to the substrate. Net wings enclosed an area 1 meter wide from 15 cm in front of the sampled habitats (to allow for gear deployment) to the inland extent of tidal flooding. Riprap rocks were removed down to the underlying filter cloth in order to place the wings in the revetment shorelines, and then rocks were placed back around the wings as found. A 50 cm high filter cloth wall was built across the back of the fringing marsh stations at site B because high tides flooded over the berm at this site into a shallow pond. Nekton movement into the flume nets from this shallow pond could have altered results if the connection was left open. Flume nets were not sampled for one week after construction to allow for the habitat inside and adjacent to the flume net station to recover from disturbance during construction. No fouling organisms, and very little sediment trapping was observed on the flume net walls during the course of the study.

Block nets, made with cotton sheets weighted with 0.8 cm steel chain at the bottom and buoyed by pipe insulation at the top, were fixed to 1.9 cm diameter schedule-40 PVC on both sides. The 1.9 cm PVC was made to slide into 2.5 cm diameter PVC pipe tracks attached to the most creekside stake of the flume net wings. The block nets were suspended by trip lines tied to stakes 10 meters offshore. Cod-end nets were made with 1.5 mm square Delta 35 lb. test nylon netting. This mesh size retained 100% of mummichogs greater than 9 mm TL during extended aquarium observation. Cod-end nets were also built to slide into a PVC collar on the front of the flume net stake while the block net remained in place. This PVC track system completely sealed deployed block and cod-end nets to the flume net sidewalls, preventing nekton escape.

Sampling

Fourteen flume nets were installed at randomly chosen locations at the three paired study sites. Three flume nets were installed into both the riprap and fringing marsh shoreline at site A, while two nets were built into each shoreline type at sites B and C. The fringing marsh flume nets at site A and B were moved to different locations along the shoreline in early August due to marsh die back inside the flume walls. Only two of the three nets at site A were rebuilt. Nekton were collected from revetment and fringe marsh flume nets at one site during each tide sampled. Nets tripped during full daylight were considered day samples and nets tripped under total darkness were considered night samples.

Sampling was conducted during new moon, spring high tide periods from May through August 2000 (Table 4). The full moon spring tide was sampled in mid-May. Full moon sampling was discontinued after May because the full moon tidal range was insufficient for flume net sampling in Sarah Creek during the summer of 2000. Spring tides were sampled because a large tidal range is necessary to drain the intertidal habitats completely when using flume nets, and because fundulid spawning cycle in marshes is correlated to cyclical spring tide events (Abraham 1985). The sampling schedule (Table 4) reflects the order that flume net construction was completed at the three sites. Site A and B were not sampled during the 5^{th} sampling period because the flume net stations had to be relocated due to marsh die-back within the original sampling area. The order that paired sites were sampled during each sampling period was chosen randomly. Tidal predictions and lunar cycles were based on predictions for Gloucester Point, VA by the Tides and Currents for Windows program, Nautical Software®, Inc.

This study encompassed the time period of fundulid recruitment and greatest nekton utilization of salt marsh habitats in Virginia. Sampling started in May because the largest recruitment of juvenile fish into the shallows at Goodwin Island, Va. occurred between mid-May and early June in 1996 (G. Cicchetti personal communication). *Fundulus heteroclitus* (mummichog) post larvae and small juvenile abundance peaked in June in saltmarshes on the Eastern Shore of Virginia (Yozzo et al. 1994). Sampling ended in early September. The last sampling period took place during the period of maximum *Callinectes sapidus* (blue crab), mummichog, and *Palaemonetes pugio* (grass shrimp) abundance in Goodwin Island intertidal areas in 1998 (Cicchetti 1998a). Total fish abundance was highest on salt marshes on the Eastern Shore of Virginia between June and August (Yozzo et al. 1994).

Flume net wings were inspected for holes and escape routes before every sampling. Block nets were hung at the low tide prior to the sampled high tide. It was assumed that nekton gained access to the flooded intertidal habitats and utilized the habitat inside the flume net wings normally as the tide rose (McIvor and Odum 1986). The block nets were remotely tripped in random order as close to slack high tide as possible, trapping nekton in the riprap or salt marsh area within the net wings. Nekton abundance and diversity in flooded salt marsh has been shown to be greatest at slack high tide (Kneib and Wagner 1994). The flume net could then be approached and the cod-end net slid into place (Figure 3). The chain on the cod-end net apron was pushed firmly into the substrate, with care taken to insure that there were no escape routes at the bottom corners. The cod-end net was pulled tight and staked, forming a triangular funnel. The block net was then removed, allowing nekton to passively funnel into the cod end as the tide fell, completely draining the fringe marsh or riprap intertidal area.

Cod-ends were collected as soon as the tide fell below the mouth of the net in order to limit crab tears (from both inside and outside the net) and *in situ* predation. Visual and manual inspection was performed inside the flume net area to check for stranded animals. The cod-end was then slid out of the PVC collar and placed in water-filled buckets. Nekton from each net was washed down into a bucket, sieved through 1.5-mm mesh and measured to the nearest mm. Cod-ends and the sieves were inspected under high intensity light for small nekton.

Nekton Handling and Measurement

Animals were processed as quickly as possible. Total length (TL) was measured for all fish except Atlantic silversides where fork length (FL) was recorded. Point-to-point carapace width (CW) was measured for blue crab. All fish < 20mm were preserved in 5% formalin for identification in the lab. All infrequently captured fish were preserved in 5% formalin and weighed using an Acculab V-200 scale. Fish were identified using Murdy et al. (1997) and Hardy (1978). The vast majority of fish and blue crabs were measured quickly and released alive near the study sites. Length-weight regressions used to obtain wet weights from nekton measurements are listed in Table 5. Dry weight : wet weight ratios from Thayer et al. (1973) were used to calculate nekton grams dry weight (gdw). Values for the closest morphological match were used for fish species not reported in Thayer et al. (1973).

All grass shrimp were frozen. Later, grass shrimp were thawed, identified, counted and total length (tip of rostrum to end of telson) was measured for 25 randomly selected individuals per net. Total length was used to calculate grams dry weight. The average grass shrimp gdw of the 25 grass shrimp measured from each net was multiplied by the total number of grass shrimp in that net to obtain total grass shrimp biomass.

Nekton abundance and biomass values are reported in two ways. First, results are listed per linear meter of shoreline, as this is the width between the flume net wings (McIvor and Odum 1986, Hettler 1989, Peterson and Turner 1994). These results report the total number of individuals that were in the intertidal habitat along one meter of shoreline. Total catch from each flume net sample was also divided by the tidal flood distance into each flume net to obtain nekton abundance and biomass per square meter values. These per square meter values reflect an average density of nekton inside the flooded area of the flume net when the block net was dropped. They are corrected for differences in the amount of habitat flooded between sample tides and between the fringe marsh and riprap intertidal areas. Nekton abundance was not quantified by volume of water inside the flume net, because grass shrimp, blue crabs and killifish, the dominant nekton in shallow water habitats in Sarah Creek, are demersal when in intertidal habitat; thus abundance relative to the amount of area flooded is more appropriate (Varnell and Havens 1995).

Recovery Efficiency

Estimates of flume net recovery efficiency for mummichog, *Paralichthys dentatus* (summer flounder) and blue crabs were investigated. Mummichogs were used because they are highly mobile and were the dominant fish species. Blue crabs were used because they are also highly mobile, and displayed energetic escape attempts. Summer flounder were used because they are a benthic fish species. Recovery efficiency was investigated by placing marked organisms within the size range normally captured on the marsh surface into flume nets after the cod-end had been installed during normal sampling (McIvor and Odum 1986). Mummichogs and summer flounder between 30 mm and 90 mm TL were marked by injecting a small amount of non-toxic acrylic paint under the surface of the skin at the forward base of the dorsal fin (Lotrich and Meredith 1974). Methods used for the mark-recapture study were approved by the College of William and Mary Research on Animal Subjects Committee (project 0008, 2000) and conformed to Guidelines for Use of Fishes in Field Research (ASIH 1988). Recovery efficiency trials using adult mummichogs were performed at every site to investigate potential differences in capture efficiency between sites. Recovery efficiency trials were performed over time at site A to investigate changes in recovery efficiency as the flume net installations aged.

Two methods were used to estimate recovery efficiency from the flume nets for small mummichogs whose small size and sensitivity precluded the use of physical tagging and manual handling. Post larval and juvenile mummichogs (10-25 mm TL) were captured with a fine mesh aquarium dip net from pits and depressions in the intertidal area of Sarah Creek. In the first method, fish were soaked for 15 minutes in creek water stained with Rose of Bengal and then carefully dumped into flume nets during sampling. Small mummichogs captured in the sample were immediately examined under a dissecting scope for stain marks. Juvenile fish stored in the laboratory did show some degree of stain loss when placed in unstained water. This method suggests minimum recovery efficiency because of stain loss and potential predation in the net. In the second method, cod-ends were installed on the flume nets at low tide to keep nekton out of the flume net enclosure as the tide rose. Between 50 to 66 small (10-25 mm TL) mummichogs were carefully placed, unmarked, into the flume nets at high tide. Holes in the cod-end nets were not observed prior to or after these trials, thus all nekton captured in the cod-end were assumed to be the unmarked fish placed in the flume nets.

Blue crabs were marked with a Sharpie permanent marker on the bottom carapace after the crabs were dried with paper towels. Blue crabs with very white bottom shells were used for marking in hopes that they had recently shed and would therefore not shed the mark while in the flume net.

Marked mummichogs and blue crabs observed in aquariums in the laboratory maintained clearly visible marks for at least 3 days, after which time they were released. Marking had no obvious ill effects on fish or crab health during the three-day trial period or during the mark-recapture trials. Grass shrimp recovery efficiency was not investigated due to their less mobile nature.

Data Analysis

Calculation of Means

Abundance and biomass per linear meter and per square meter data from fringe marsh and riprap nets were averaged for each tide sampled. These means (N = 31) were pooled for statistical comparison (Figure 4), are listed in statistical test tables, and are also used in the text for descriptive purposes. Standard error given in all tables with statistical results reflect variation between the 31 tides sampled. Mean values in tables are not connected to the nonparametric test results, as these tests analyze ranks, not mean values. Grand means were also calculated for each time period and are used to describe temporal trends in nekton abundance and biomass in the text (Figure 4). Sample sites are represented equally in sample period grand means. An overall mean was calculated from the six sample period grand means.

The Index of Relative Importance (Austin et al. 1996) was calculated by multiplying the percent occurrence of individual species in sample nets (# of flume nets with species captured / total flume nets) by the relative abundance of that species (total number of a species captured / total number of fish captured).

Statistical Methods

Sample tide means are the unit of replication except where noted. The Anderson-Darling Normality Test and Levene's Test for homogeneity of variance were calculated for all compared results. Nonparametric statistics were used throughout the study when normality or homogeneity of variance was not met ($\propto = 0.05$). The null hypothesis for all statistical tests was that there is no difference in nekton abundance, biomass or size between treatments. Significance levels for all tests were taken at the $\propto = 0.05$ level. Only species with more than 25 individuals captured in either marsh or riprap samples were analyzed for site to site, diel or habitat statistical comparisons.

Physical site characteristics

Paired two-sample t-test for means was used to test for differences in tidal height, intertidal distance flooded, temperature, salinity, dissolved oxygen and turbidity between fringe marsh and revetment sampling areas.

Recovery Efficiency

One-way analysis of variance was performed on recapture results from marsh and riprap trials to examine potential differences in recovery efficiency between sites (factor = site; levels = sites (A,B,C); response variable = recapture efficiency). Two-sample t-tests were used to elucidate differences in recovery efficiency between marsh and riprap and between night and day trials at site A because the most recovery efficiency trials were performed at this site.

Site Comparison

Fringe marsh and riprap site comparisons were performed in order to verify that habitat comparison results (N = 31) were not unfairly biased by site A, which was sampled more times (N = 14), than site B (N = 8) or C (N = 9). One-way analysis of variance and the nonparametric Kruskal-Wallis Test were used to compare mean nekton abundance and biomass between the three fringe marsh and three riprap sites. Pooled sample tide means from each site were used as the unit of replication (N site A = 14, B = 8, C = 9). Tukey's multiple comparison test was used to elucidate differences between sites ($q = _{0.05, 28, 3}$) when significance was observed. Tests were run on both nekton abundance and biomass per linear meter of shoreline ($\cdot$ m^{-1}) and per square meter of flooded intertidal ($\cdot$ m^{-2}). Summer flounder results were not tested as 93% (92 of the 98 fish caught) of this species were caught during the first sampling period when only one site was sampled.

Diel Use Comparison

Two-sample t-tests and the Mann Whitney tests were used to test for significant differences between day and night use of marsh and riprap habitats. The unit of replication used for the diel study was mean abundance and biomass values from marsh and riprap flume nets from eleven night-time sample tide means and eleven corresponding daytime sample tide means that were closest to the night sample date. Summer flounder data were analyzed using results from individual nets (not sample means) from the 1st sampling period because 93% of summer flounder were captured during the first sampling period. Species with greater than 25 individuals captured in either the 11 day or 11 night sampled tides were analyzed.

Habitat Comparison

Results were analyzed using abundance and biomass per meter of shoreline results (total nekton captured in each flume net) and abundance and biomass per square meter of flooded intertidal habitat. Greater nekton abundance and density in a habitat is indicative of greater habitat quality (Able 1999). This assumption can be made because habitat selection is defined as the non-random use of space due to the voluntary movements of organisms stemming from evolved responses to environmental stimuli and behavioral choices among alternative habitats (Craig and Crowder 2000). Biomass was used in the habitat comparison because this parameter incorporates both abundance and size. Total biomass of nekton recruiting to adult populations is the most important measure of nursery and juvenile habitat (Beck et al. 2001).

The unit of replication in habitat use comparison statistics is mean abundance and biomass values from marsh and riprap flume nets per sample tide (N = 31). The Wilcoxon paired rank sign test was used to examine differences in nekton mean abundance and mean biomass between fringe marsh and riprap habitat (Beauchamp et al. 1994). Samples can be considered paired because fringe marsh and riprap flume nets at each site were always sampled on the same tide (Zimmerman and Minello 1984, Kneib and Wagner 1994, Howe et al. 1999). Paired sample means were pooled for analysis (West and Zedler 2000). Individual site results were also tested (N site A =14, B = 8, C = 9) for species when differences between sites were found. Sample means were tested for day (N = 20) and night (N = 11) results when significant diel differences were found. Separate site and day/night tests were performed to check if uneven representation was driving pooled data results. The Mann-Whitney Test was used for summer flounder data obtained from individual nets during the first sampling period only at site A (pooled n = 17, day n = 8, night n = 9). A non-paired test was used for summer flounder results because data from individual nets cannot be considered paired.

Determination of Intertidal Area Dependency

Total abundance of mummichog, blue crab and grass shrimp from individual nets during the May 4-8 sampling at site A were plotted against flood distance inside the nets to investigate the relationship between flooded intertidal area and nekton abundance (N = 17 individual nets in both fringe marsh and riprap). Riprap and fringe marsh nets were analyzed separately. Data from the 1st sampling period were used because the most samples at any one site during a sampling period were collected during this time. A significant positive relationship between abundance and distance flooded would be expected for species categorized as intertidal area dependent. Intertidal area independent species would be expected to display no trend in abundance with greater intertidal flood distance.

Length

Mummichog, grass shrimp and naked goby (*Gobiosoma bosc*) total length, Atlantic silverside (*Menidia menidia*) fork length and blue crab carapace width was compared between fringe marsh and riprap for each sample period using the Mann-Whitney test. All individuals captured during each sample period were pooled for comparison (West and Zedler 2000). Tests were performed only when more than five individuals were captured from both the fringe marsh and the revetment during that period. The Wilcoxon paired sign rank test was used to compare median size of mummichog, blue crab and grass shrimp data from the 31 sample tides. Mann-Whitney tests using pooled data from all individuals captured during the entire study were also performed for naked goby and Atlantic silverside. Statistical tests were not performed on summer flounder, striped killifish (*Fundulus majalis*), white perch (*Morone americana*), spot (*Leiostomus xanthurus*), etc. because too few of each species was captured either in the marsh or riprap for comparison. Length-frequency histograms per sampling period were made for mummichog, blue crab, naked goby, Atlantic silverside and grass shrimp data.

RESULTS

Site Description

Fringe marsh and riprap site characteristics are described in Tables 2 and 3. The mean offshore slope was 5% and 4.7% in front of fringe marsh and revetment sites respectively. The slope of the intertidal fringe marsh averaged 17.6%. The average slope of the ground under the revetment rocks was 30.4%. The slope of the surface of the revetment rocks in the intertidal area averaged 45.7%. Sediment from the edge of fringe marsh stations was 0.3% gravel, 86.9% sand, 5.4% silt and 7.5% clay on average. Sediments from the edge of the riprap sites averaged 0.5% gravel, 89.8% sand, 3.1% silt and 6.6% clay.

The average difference in salinity, dissolved oxygen, water temperature and turbidity values between paired fringe marsh and riprap sample tides was < 0.1, < 0.1 mg · L^{-1}, 0.4° C and 2.0 NTU respectively. The maximum variation between paired fringe marsh and riprap sites for these parameters during the study was 3.0, 0.9 mg · L^{-1}, 2.3 °C and 7 NTU. Salinity and dissolved oxygen were not significantly different between the fringe marsh and riprap sample areas (Table 6). Water temperature and turbidity were statistically greater in fringe marsh stations. However, the actual values (0.4°C warmer and 2.0 NTU's higher in fringe marsh) were minimal and therefore can be considered negligible (Table 6). The average temperature and salinity during the six sampling periods varied between 20.9° C to 28.5° C and 13.1 to 19.7. The average dissolved oxygen levels and turbidity values during the six sampling periods varied between 5.5 mg · L^{-1} to 7.8 mg · L^{-1} and 13 NTU to 19 NTU during the study period. The lowest oxygen condition observed during sampling was 4.1 mg · L^{-1}.

Although the order that nets were tripped was randomly picked, riprap block nets were tripped when the tidal height was 2.0 cm greater at the intertidal habitat edge on average than fringing marsh flume nets. Although this difference was statistically significant (Table 6), the small disparity in tide height can be considered negligible (Figure 5). Tidal heights at the time the block nets were tripped in fringe marsh were between 36 and 76 cm (average = 57 cm, SE = 1.8). Tidal heights for riprap samples were between 37 and 69 cm (average = 59 cm, SE = 1.6).

The tidally flooded area was 1.1 m^2 greater on average in fringe marsh stations (Figure 6). The mean intertidal area flooded inside fringe marsh flume nets over the course of the study was 3.0 m^2 (SE = 0.1) with a range of 1.7 m^2 to 3.7 m^2. For the revetment sites, mean intertidal area flooded was 1.9 m^2 (SE = 0.1) with a range of 1.4 m^2 to 2.4 m^2. The fringe marsh at site A had a significantly greater mean flooding distance (3.3 m^2, SE = 0.1) than site B (2.7 m^2, SE < 0.1, Table 6). The riprap flume nets at site B flooded to a statistically greater extent (2.4 m^2, SE < 0.1) than sites A (1.8 m^2, SE = 0.1) and C (1.7 m^2, SE < 0.1, Table 7). Site A had the largest disparity in flooded intertidal area between fringe marsh and riprap flume nets.

Recovery Efficiency

Mummichog

Recovery efficiencies for adult mummichog (35 – 90 mm TL) were high throughout the study (Figure 7). Eighty-eight percent of large mummichog were recaptured from fringing marsh flume nets, and 89% of marked fish were recaptured from riprap flume nets (Table 8). Mummichog recovery efficiency was not significantly different between the three sampling sites for flume nets placed in natural fringing marshes or riprap (Table 9), or between fringe marsh and riprap flume nets (Table 10), or day and night trials (Table 10), or during the first 10 weeks of the study.

Recovery efficiencies of small mummichog (10 – 25 mm TL) were high with both methods (Table 8). Almost eighty-one percent of fish stained with Rose Bengal held the mark and were recaptured during the two trials. Recovery of small mummichog was also high in the trials using unmarked fish placed in flume nets that were void of other nekton. Ninety-five percent of the small mummichogs were recovered from fringe marsh flume nets and 68% were recovered from riprap flume nets during these trials.

Blue Crab

Eighty-three percent of marked blue crabs (25 – 60 mm CW) were recaptured during two fringe marsh mark-recapture trials. Thirty-eight percent of marked blue crabs were recaptured during two riprap flume net trials. Recovery efficiencies during the 2 trials (10 marked blue crabs were used each time) were 76% and 0%.

Summer Flounder

100% (4/4) marked juvenile summer flounder (40-60mm TL) placed in a fringe marsh flume net were recaptured. Summer flounder were often found half eaten in the cod ends throughout the study, suggesting that they were susceptible to higher predation rates once trapped inside the flume net.

Habitat Use: Abundance

A total of 141 successful flume net samples (71 fringe marsh and 70 revetment) from 20 daytime tides and 11 night tides were obtained over the course of the study. Flume nets captured nekton from 8 mm TL (mummichog and naked goby) to 445 mm TL (American eel, *Anguilla rostrata*); flume nets caught nekton that were generally less than 100 mm TL. Numerically, grass shrimp comprised 90.1%, fish comprised 6.5% and blue crabs comprised 3.4% of the total individuals captured. Fringe marsh flume nets samples contained a total of 1,432 fish, 765 blue crabs and 14,354 grass shrimp (Table 11). Riprap flume net samples contained a total of 801 fish, 385 blue crabs and 16,414 grass shrimp. Fish, blue crabs and grass shrimp were analyzed separately due to the overwhelming abundance of grass shrimp in this study.

Populations of blue crabs (Figure 12a, 13b), summer flounder, Atlantic silversides (Figure 13e), white perch and spot captured in this study were dominated by young of the year (YOY) and juvenile life-history stages (Table 25). The majority of mummichog (Figure 12b, 13c) and grass shrimp (Figure 13a) captured were of sexually mature size.

Catches of naked goby (Figure 12c, 13d) and striped killifish (Table 25) were more evenly split between juvenile and mature individuals.

A total of 21 species (19 fish and 2 crustaceans) were captured (Table 11). Eighteen species were captured in fringe marsh flume nets and 13 species were captured in riprap samples. Blackcheek tonguefish (*Symphurus plagiusa*, 11 individuals), white mullet (*Mugil curema*, 6 inds), striped bass (*Morone saxatilis*, 2 inds), rainwater killifish (*Lucania parva*, 2 inds), *Bairdiella chrysoura* (silver perch, 2 inds), bluefish (*Pomatomus saltatrix*, 1 ind), spadefish (*Chaetodipterus faber*, 1 ind) and one *clupeid* sp. were species captured only in fringe marsh. Skilletfish (*Gobiesox strumosus*, 8 individuals), striped blenny (*Chasmodes bosquianus*, 2 inds), American eel (1 ind) and a penaid shrimp were species only captured in riprap flume net samples.

The bulk of total abundance for each fish species was captured in the fringe marsh (Table 11, Figure 8). Grass shrimp and 'other fish' were captured in similar numbers in both habitats. Naked goby was the only species captured in greater numbers in riprap shorelines. A greater number of fish species were captured more frequently (% occurrence) in the fringe marsh (Table 11). Relative abundance (% abundance) of fish species is distributed among more species in the fringe marsh, as can be observed by the shallower slope of the fringe marsh abundance curve as compared to the curve for riprap (Figure 10). Seven species of fish with greater than 1% of relative abundance in the fringe marsh comprised 97.6% of the total fish captured in fringing marsh. These were mummichog (66.0%), naked goby (8.9%), Atlantic silverside (7.1%), summer flounder (6.4%), striped killifish (5.6%), white perch (2.1%) and spot (1.7%). Three species of fish with greater than 1% of relative fish abundance in the riprap comprised 95.3% of total fish captured in the riprap (Table 11). These were mummichog (64.2%), naked goby (25.7%) and Atlantic silversides (5.4%). In both fringe marsh and riprap, the relative abundance of fish species in either habitat and that species contribution to the Index of Relative Abundance (IRI) is similar, indicating that infrequent capture of large numbers of a species did not occur.

The majority of fish species considered to be commercially or recreationally important (92.4% or 159 out of 172 total individuals) were captured in fringe marsh flume net samples (Figure 8). This includes 92.9% (91/98) of summer flounder, 93.8% (30/32) of white perch, 96.2% (25/26) of spot and 100% of white mullet (6/6), striped bass (2/2), silver perch (2/2), bluefish (1/1) and spadefish (1/1). In comparison, 2/3 seatrout and 1/1 American eel were caught in riprap.

Significantly greater mean abundances of mummichogs, Atlantic silversides, summer flounder, striped killifish, white perch, spot, total fish and blue crabs per linear meter of intertidal habitat edge were found in fringe marsh shoreline (Table 18 and 20). There were no significant differences in naked goby and grass shrimp abundance per meter between riprap and fringe marsh shorelines (Table 18).

Significantly greater density per square meter of summer flounder, striped killifish, white perch, spot and blue crabs were captured per square meter of flooded fringe marsh (Table 18 and 20). No difference in mummichog, Atlantic silversides or total fish density per square meter was found. Significantly greater density per square meter of naked goby and grass shrimp were captured in flooded riprap (Table 18).

No relationships between grass shrimp, blue crab or mummichog abundance and intertidal area flooded were found during the first sampling period at site A.

Habitat Use: Biomass

Blue crabs were the dominant nekton in terms of total nekton biomass in both fringe marsh (44.1%) and riprap (44.9%), followed closely by grass shrimp biomass (34.0% of fringe marsh total biomass, 39.7% of riprap total biomass, Table 12). Fish comprised 21.9% of the total nekton biomass captured in the fringe marsh samples and 15.4% of the total nekton biomass captured in the riprap.

Fish biomass was distributed among more species in the fringe marsh than in the riprap (Figure 11). Seven fish species with greater than 1% of total fringe marsh biomass made up 94.8% of the total fish biomass captured in the fringe marsh. These were mummichog (74.2%), striped killifish (10.8%), summer flounder (4.0%), spot (3.0%), Atlantic silverside (1.9%), naked goby (1.6%) and white mullet (1.3%, Table 12). Three species of fish with greater than 1% of the biomass in riprap samples comprised 98.6% of total fish biomass captured in riprap (Table 12, Fig 11). These were mummichog (90.8%), naked goby (6.8%) and Atlantic silverside (1.1%). Mummichogs played a more significant role in riprap fish relative biomass (90.8%) compared to the relative abundance of mummichogs in riprap (64.2%). Alternately, naked goby played a less significant role in riprap relative biomass (6.8%) compared to the relative abundance of naked goby in the riprap (25.7%).

The bulk (97.2%) of commercially important fish biomass was captured in the fringe marsh (Figure 9). This includes 94.7% of the total summer flounder biomass, 95.0% of total white perch biomass and 100% of the total spot biomass. Total biomass of grass shrimp, naked goby, striped blenny and skilletfish was greater in the riprap. Total biomass of blue crabs and all other fish species was greater in the fringe marsh (Table 12, Figure 9).

Biomass of Atlantic silversides, summer flounder, striped killifish, spot and total fish was significantly greater per linear meter of shoreline in the fringe marsh (Tables 19 and 20). Naked goby biomass per meter was significantly greater in riprap shorelines. Differences in biomass per meter of riprap and fringe edge was not observed for mummichog, white perch, blue crab and grass shrimp.

Biomass per square meter values for summer flounder, striped killifish and spot were greater in fringe marsh (Tables 19 and 20). Significantly greater biomass per square meter of mummichog, naked goby and grass shrimp were captured in the riprap. Differences in biomass per square meter of flooded riprap and fringe marsh were not observed for Atlantic silverside, white perch, total fish and blue crab.

Intra Habitat Comparison

Significantly greater abundance and density of grass shrimp were captured in site A fringe marsh versus the site C fringe marsh (Table 14). Mean grass shrimp biomass was also greater in the fringe marsh at site A than site C, consistent with the trend in abundance (Table 15). Blue crabs were significantly more abundant per square meter in fringing marshes at site B than site A but there was no significant difference in blue crab biomass between fringe marsh sites (Table 14).

Significantly greater numbers of blue crabs were captured in the riprap at site B than at riprap sites A and C (Table 14). Blue crab biomass values were not significantly different among the three riprap sites (Table 15). Naked goby abundance and density in riprap flume net samples were statistically greater at site C than site A (Table 14). Naked goby biomass trends were similar to abundance observations (Table 15).

Diel Habitat Use

Significantly greater numbers of Atlantic silversides and summer flounder were captured in the fringe marsh at night on both a per linear meter of shoreline and a per square meter basis (Table 16). Almost all (84 out of 85) individual summer flounder captured from the marsh surface during the 1st sampling period were captured at night. A similar pattern was observed for Atlantic silversides, where 77 out of 92 (83.7%) total individuals from the samples used for diel analysis were captured in the marsh at night. Biomass values for Atlantic silversides and summer flounder were correspondingly greater in the fringe marsh at night (Table 17). Too few Atlantic silverside and summer flounder were captured in the samples used for the diel comparison to test for diel differences in utilization of intertidal riprap, however diel trends in habitat use for both Atlantic silversides and summer flounder was similar in the riprap.

Blue crab biomass per square meter in the fringe marsh was significantly greater during the day (Table 17) although the density of blue crabs did not show a significant diel trend in the fringe marsh (Table 16). Mummichog and total fish biomass were also significantly greater during the daytime in riprap samples (Table 17) while diel differences were not found for mummichog or total fish abundance in the riprap (Table 16).

Total Fish and Species Specific Results

Total Fish

Total fish abundance peaked in both habitats during August (Table 13). Abundance of all fish combined averaged 19.9 (SE = 2.3) individuals per meter of fringe marsh shoreline (6.4 inds · m^{-2}, SE = 0.6) verses 11.7 (SE = 1.4) individuals per meter of riprap shoreline (6.1 inds · m^{-2}, SE = 0.8, Table 18). Total fish abundance per meter was significantly greater in fringe marsh shoreline, while no difference was found in total fish density per square meter (Table 18).

Total fish biomass per linear meter of fringe marsh shoreline (11.36 gdw · m^{-1}, SE = 1.22) was significantly greater than the riprap shoreline (7.84 gdw · m^{-1}, SE = 0.80, Table 19). Significant difference in total fish biomass per square meter between fringe marsh and riprap was not observed for pooled day and night results. However, total fish biomass per square meter was significantly greater in the riprap during the day, when riprap values (4.83 gdw · m^{-2}, SE = 0.43) were greater than fringe marsh values (3.76 gdw · m^{-2}, SE = 0.43, Table 19). This reflects the pattern observed for mummichog biomass.

Commercially Important Fish

The commercially important fish category includes all species of fish that have commercial or recreation value either in the Chesapeake Bay, or on the East Coast. Summer flounder, white perch, spot, white mullet, striped bass, silver perch, seatrout, bluefish, spadefish and the American eel are represented. Abundance of these fish was greatest during the 1st sampling period in the fringe marsh (5.35 inds · m^{-1}, SE = 2.0; 1.38 inds · m^{-2}, SE = 0.6) when the majority of summer flounder were captured. Commercially important fish abundance averaged 1.8 inds · m^{-1} (SE = 0.59) in the fringe marsh (0.5 inds · m^{-2}, SE = 0.16) and 0.2 inds · m^{-1} (SE = 0.05) in the riprap (0.09 inds · m^{-2}, SE = 0.03) during the six sampling periods.

Grass Shrimp

Palaemonetes pugio was the only species of grass shrimp and the most abundant species captured in this study (n = 30,768 individuals). Abundance peaked during the August 27 – September 2 sampling period, while the greatest biomass values were observed in mid-summer (Table 13). Adult grass shrimp (> 20mm TL) comprised a large majority of the population during the first four sampling periods, while young of the year (YOY) played a large role during August. Length frequency histograms show that a cohort of mature grass shrimp approximately 30 mm TL was captured during May 4-8 (Figure 13a). These shrimp grew to approximately 40 mm TL by June 28-July 5. Young of the year grass shrimp were captured in large numbers beginning in the June 28-July 5 sampling period. This cohort grew to approximately 25 mm TL and became the dominant portion of the grass shrimp population by the end of August.

Statistically similar numbers of grass shrimp were captured per meter of shoreline in the fringe marsh (197.7 inds · m^{-1}, SE = 22.4) and riprap (231.8 inds · m^{-1}, SE = 22.3, Table 18). Grass shrimp biomass per meter was also similar in fringe marsh (15.87 gdw · m^{-1}, SE = 1.72) and riprap (17.72 gdw · m^{-1}, SE = 1.74, Table 19).

Significantly greater abundances of grass shrimp per square meter were captured in riprap habitat (122.5 inds · m^{-2}, SE = 11.8) compared to fringe marsh (67.4 inds · m^{-2}, SE = 6.9, Table 18). Grass shrimp biomass per square meter was also significantly greater in riprap (9.48 gdw · m^{-2}, SE = 0.97) than fringe marsh (5.46 gdw · m^{-2}, SE = 0.57, Table 19).

Results from Wilcoxon paired rank sum test comparing median grass shrimp length from the 31 paired samples were not significant (Table 24c). While significant differences in mean total length were observed during four sample periods, no overall trend was observed and the differences are small considering that animals were measured to the nearest millimeter. This evidence points to the fact that grass shrimp sizes were comparable between the two habitats.

Blue Crab

Blue crabs were the biomass dominant species and the second ranked species in abundance, following grass shrimp. Blue crab abundance peaked during the August 27-September 2 sample period due to an influx of crabs between 20-30 mm CW (Table 13, Figure 13b). Fringe marsh biomass values also peaked during this time, while riprap biomass was greatest during May 16-18. The vast majority (91.1%) of blue crabs were smaller than 70 mm CW and are considered juveniles (Dittel et al. 1995). Crabs between 20-40 mm CW dominated catches from June 1st until September 2nd (Figure 13b).

The 0% recovery efficiency observed during one of the two recovery efficiency trials suggests that blue crabs may have avoided capture more successfully in riprap flume nets, however this result is tempered by the low replication of the blue crab recovery efficiency experiment. Therefore, it is possible that observed abundance and biomass trends do not reflect reality. Results, not considering recovery efficiency, show that significantly more blue crabs were captured per linear meter of shoreline in the fringe marsh (10.9 inds $\cdot$ m^{-1}, SE = 1.1) compared to riprap (5.7 inds $\cdot$ m^{-1}, SE = 0.8, Table 18). Blue crab density was also significantly greater in the fringe marsh (4.0 inds $\cdot$ m^{-2}, SE = 0.4) compared to riprap (2.9 inds $\cdot$ m^{-2}, SE = 0.4). Although approximately two times as many blue crabs were captured per meter of fringe marsh shoreline, biomass per meter of edge values between fringe marsh and riprap were not significantly different (Table 19). Similarly, blue crab biomass per square meter in the fringe marsh (8.03 gdw $\cdot$ m^{-2}, SE = 0.96) was not significantly different from blue crab biomass per square meter in the riprap (12.10 gdw $\cdot$ m^{-2}, SE = 1.70, Table 19) even though significantly more individuals were captured per square meter in the fringe marsh.

Eighty-five percent of the total number of crabs captured in the fringe marsh were smaller than 50 mm CW, compared to 65% in the riprap (Figure 12a). Juvenile crabs between 20-50 mm CW were caught in greater numbers in the fringe marsh during the entire study (Figure 13b). One hundred and forty one percent more small crabs (<25 mm CW) were caught in the fringe marsh compared to riprap samples. Crabs greater than 60 mm TL were captured to a similar degree in either habitat (Figure 13b), although larger crabs contributed a greater proportion to blue crab relative abundance in the riprap (Figure 12a). Median blue crab carapace width was 9 mm larger (37%) in the riprap (Table 24b). Mean blue crab carapace width was significantly greater in riprap during 5 out of the 6 sampling periods. The difference in blue crab carapace width between fringe marsh and riprap ranged from 5 mm during May 4-8 to 28 mm during Aug 1-2.

Blue crab biomass per square meter in fringing marsh was significantly greater during the day (Table 17), while blue crab density per square meter in the marsh was similar between day and night (Table 16). This indicates that larger blue crabs were found in the fringing marsh during the day. These results are driven by the capture of very large crabs in the fringe marsh during the day rather than by differences in median size. Nine crabs greater than 100 mm CW were captured in daytime marsh samples, compared to only 1 at night. These crabs exert a large influence on biomass values. The mean length of blue crabs in the fringe marsh did not differ between day and night.

Mummichog

Mummichog were the most abundant fish captured in both the fringe marsh and riprap. Mummichog were captured in 95.8% and 98.6% of individual fringe marsh and riprap flume net samples, respectively (Table 11). Mummichog relative abundance (66.0% and 64.2% in fringe marsh and riprap) and their contribution to the Index of Relative Importance in both the fringe marsh and riprap (6,323 and 6,330) are almost identical. Mummichog abundance and biomass were high throughout the study (Table 13). The majority (69% in fringe marsh, 74% in riprap) of mummichog captured were mature (> 30 mm TL) individuals (Figure 12b).

Larval mummichog (< 25 mm TL) are a significant component of the population from the May 16 – 18 sample period through the August 27 – September 2 sample period. Length frequency plots (Figure 13c) show a cohort of larval mummichog appearing during the May 16-18 sampling period. This cohort of mummichog grew to between 20 – 25 mm TL by the end of June. Mummichog grow to 30-35 mm TL in 5-6 months (Abraham 1985), therefore the peak of 50-55 mm TL individuals seen during the last sampling period were not this YOY cohort. These 50-55 mm TL fish were likely mummichog spawned in the previous year. A cohort of larger mummichog (approximately 55 mm TL) was observed in early May and grew to a length of approximately 65 mm TL during June 28-July 5 before the distribution lost its bimodal nature (Figure 13c). One hundred and twenty two percent more small mummichog (< 25 mm TL) were captured in the fringe marsh compared to the riprap.

Mummichog abundance per meter of shoreline was significantly greater in the fringe marsh (13.0 inds $\cdot$ m^{-1}, SE = 1.7) compared to riprap (7.2 inds $\cdot$ m^{-1}, SE = 0.8, Table 18). No difference in mummichog density per square meter between marsh (4.3 inds $\cdot$ m^{-2}, SE = 0.5) and riprap (3.8 inds $\cdot$ m^{-2}, SE = 0.4) was observed (Table 18).

Mummichog total length was significantly larger in riprap samples compared to fringe marsh samples during five out of six sampling periods (Table 24a). Median mummichog total length was 9 mm greater (18% larger) in riprap samples. The difference in median total lengths was lowest (2 mm larger in riprap) during the 1st sampling period (May 4-8) and greatest (24 mm larger in riprap) during the 4th sampling period (June 28-July 05). Forty-five percent of the mummichog captured in the riprap were greater than 60 mm TL, compared to 26% in the fringe marsh (Figure 12b).

Mummichog biomass per linear meter of shoreline was not significantly different between the two sampled habitats (fringe marsh = 8.39 gdw $\cdot$ m^{-1}, SE = 0.96; riprap = 7.05 gdw $\cdot$ m^{-1}, SE = 0.84, Table 19) although significantly greater abundance per meter of mummichog was captured in the fringe marsh. Mummichog biomass per square meter was significantly greater in the riprap (3.68 gdw $\cdot$ m^{-2}, SE = 0.39) compared to fringe marsh (2.81 gdw $\cdot$ m^{-2}, SE = 0.30), although a significant difference in mummichog density per square meter was not observed. These biomass results are most likely due to the presence of larger mummichog in the riprap during the day. Mean mummichog length was 12 mm, or 28.3% larger, during the day in the riprap compared to the fringe marsh. There was no size or biomass difference of mummichog at night.

Naked Goby

Naked gobies were the second most abundant fish captured, comprising 8.9% of the total fish captured in the fringe marsh and 25.7% of fish captured in the riprap. Naked gobies accounted for 15.6% of the total IRI in riprap compared to 3.0% of the fringe marsh total IRI . Naked goby contribution to total biomass was less than this species' contribution to abundance, comprising only 1.6% of the total biomass of fringe marsh fish and 6.8% of fish biomass in riprap (Table 12). Fifty-two percent of the naked gobies captured in the fringe marsh and 32% of gobies captured in riprap were < 25 mm TL and are considered to be juveniles (Figure 12c, Hardy 1978). Median naked goby TL was 5 mm larger in the riprap (Table 24d).

Naked goby abundance and biomass was low in both intertidal habitats during most of the study before rising sharply during the last two sample periods (Table 13). The majority of naked gobies captured during the early summer were caught in riprap stations. A pulse of juvenile and adult naked gobies was observed in both habitats during the August 1-2 sample period (Figure 13d). These fish were still apparent during the last sampling period (August 27 - September 2), when gobies of all size classes (20 - 50 mm) were captured in similar numbers in both fringe marsh and riprap.

Naked goby abundance per linear meter of shoreline was not significantly different between fringe marsh (2.1 inds · m^{-1}, SE = 0.9) and riprap (3.3 inds · m^{-1}, SE = 1.1, Table 18). Density per square meter in the riprap was significantly greater (1.8 inds · m^{-2} SE = 0.6) than fringe marsh (0.7 inds · m^{-2}, SE = 0.3, Table 18). Goby biomass was significantly greater in riprap for both linear meter and square meter results (Table 19).

Comparisons of naked goby abundance and biomass between sites reveal that naked gobies were more numerous in the riprap at site C. This site accounted for 68.1% of all gobies captured in the riprap habitats. The riprap at site C (4.6 inds · m^{-2}, SE = 1.80) was located in a more protected area than the site C fringe marsh (0.9 inds · m^{-2}, SE = 0.42, Table 18). This physical difference in the site C paired sampling area could favor nekton abundance in site C riprap as estuarine resident fish have been found in greater abundance in more protected areas (Rozas et al. 1988, Hettler 1989).

Atlantic Silversides

Atlantic silversides were the third most abundant fish captured, 70.1% of which were captured in fringe marsh flume nets. Atlantic silversides comprised 7.1% of the total fish catch in the fringe marsh and were captured in 32.3% of fringe marsh samples (3.2% of fringe marsh IRI), compared to 5.4% relative abundance and 5.7% occurrence in riprap habitat (0.4% of the riprap IRI, Table 13). Catch of silversides in the fringe marsh peaked during the August 27 – September 2 sample period (Table 13). Silverside biomass increased in the fringe marsh over the summer as silverside median fork length increased from 11 mm to 52 mm during this study (Table 24e, Figure 13e). Patterns of silverside abundance and biomass are more difficult to discern in the riprap, where 49%

of all silversides taken from the riprap (22 individuals) were captured in one net during the second sampling period (Figure 13e). Atlantic silversides grow to approximately 80 mm by the end of their first summer in the Chesapeake Bay (Holmquist 2001). Thus, all silversides captured in this study were spawned during the spring and summer. Small silversides (10-15 mm FL) were present during the May 4-8 and May 16-18 sampling periods (Figure 13e). This cohort grows to between 50-60 mm on average by the August 27 – September 2 sample period.

Silverside abundance and biomass per linear meter results were significantly greater in the fringe marsh. Silverside abundance averaged 1.4 inds $\cdot$ m^{-1} (SE = 0.4) in the fringe marsh and 0.7 inds $\cdot$ m^{-1} (SE = 0.4) in the riprap. Silverside biomass averaged 0.23 gdw $\cdot$ m^{-1} (0.12) in the fringe marsh and 0.09 gdw $\cdot$ m^{-1} (SE = 0.07) in the riprap. Differences in abundance and biomass per square meter were not observed between the two habitats. Silverside density averaged 0.5 inds $\cdot$ m^{-2} (SE = 0.1) in the fringe marsh and 0.4 inds $\cdot$ m^{-2} (SE = 0.4) in the riprap. Silversides were captured in the fringe marsh during the entire study, while they were only captured in significant numbers in riprap during the 2nd , 5th and 6th sampling periods.

The majority (76/101) of silversides captured in the fringe marsh were caught at night (Table 16). Small silversides (< 25 mm FL) were captured during both the day and the night when they were present in May and early June, however silversides larger than 25 mm FL were captured predominantly at night. Mean silverside FL was 20 mm in the fringe marsh during the day (n = 25) and 42 mm at night (n = 76). The trend of larger silversides utilizing the intertidal habitat only at night was also observed in the riprap.

Summer Flounder

Summer flounder comprised 6.4% of fish captured in fringe marsh and were the third most important fish in terms of fish biomass in the fringe marsh, even though they were predominantly captured only during the 1st sampling period. The vast majority of summer flounder were captured at night during the first sampling period (84/85 in fringe marsh, 6/7 in riprap). Summer flounder abundance per meter of fringe marsh shoreline averaged 9.33 inds $\cdot$ m^{-1} (SE = 1.5) or 2.52 inds $\cdot$ m^{-2} (SE = 0.37) during the May 4 – 8 nighttime samples. Flounder comprised 44.2% of total fish abundance in the fringe marsh during these night samples, second to mummichog which made up 46.8% of total fish abundance. Abundance in the riprap during the nighttime trials during May 4 – 8 was 0.67 inds $\cdot$ m^{-1} (0.37 inds $\cdot$ m^{-2}). Abundance and biomass per meter and per square meter were all significantly greater in the fringe marsh. The median size of fringe marsh flounder in May was 55 mm TL, with a size range of 38 – 82 mm TL in May samples. A 132 mm TL individual was captured on August 30. These fish were all YOY recruits (CBNERRVA 2001).

Striped Killifish

Striped killifish exhibited a strong preference for marsh habitat. The vast majority of striped killifish, 98% (80/82), were captured in the fringe marsh. Striped killifish were the fifth most abundant fish captured in this study, comprising 5.6% of total fish in the fringe marsh. They were the second most important fish species in terms of biomass, comprising 10.8% of total fringe marsh fish biomass. The mean size of striped killifish captured in the fringe marsh was 59 mm TL, with a range of 20 – 102 mm (Table 25).

White Perch and Spot

White perch and spot, the seventh and eighth most abundant fish captured in the study, respectively, displayed a distinct preference for fringe marsh habitat. Ninety-four percent of total white perch and 96% of total spot were captured in fringe marsh flume nets. Spot, which grow to 115 – 130 mm TL by the end of their first year and mature at 175 mm TL, were all YOY in this study. Median length of spot was 57 mm TL (Table 25). Spot ranked fourth in terms of overall fringe marsh fish biomass. White perch median length was 44.5 mm TL. White perch ranked eighth in terms of overall fringe marsh biomass. The three largest white perch (245, 245 and 260 mm TL) were captured at night. These fish were not included in biomass or total length statistics for white perch or total fish due to the exorbitant influence they would have especially on biomass values. Diel use statistics were not performed because too few individuals were captured during the sample tides used for diel statistics. However, 80% (20/25) of the spot and 83% (25/30) of the white perch captured in the fringe marsh were caught during the day.

DISCUSSION

Flume Net Recovery Efficiency

Recapture results for mummichog > 35 mm TL over the first 10 weeks of the study at site A were consistently high. This suggests that the flume net sides, block nets and cod-end nets maintained their integrity and effectiveness over time. Marked adult mummichogs were recaptured 88% of the time in this study (Table 8). This number is comparable to the 73% recovery efficiency of mummichogs (32 - 91 mm TL) from flume nets built into tidal fresh water marsh by McIvor and Odum (1986) and the 84% recovery efficiency of mummichogs (29-102 mm TL) from drop rings used by Cicchetti (1998a) to sample *Spartina alterniflora* marsh on the Goodwin Islands, Virginia. Recovery efficiency of larval mummichogs (10-25 mm TL) was between 68% and 95% over the six trials. No visible water filled pits or depressions where fish could have resided during low tide were observed inside any flume nets.

Eighty-three percent of marked blue crab were recaptured from fringe marsh flume nets during 2 trials. This recovery rate is comparable to the 86% recovery rate of larger blue crabs (50-100 mm CW) from drop rings in *Spartina alterniflora* habitats (Cicchetti 1998a). A 0% recovery rate during one of the two riprap blue crab trials reduced the average blue crab recovery efficiency in riprap to 38%. In order for blue crabs to avoid capture in the flume nets, they would have to strand themselves, climb up and over the cod-end mesh or riprap rocks, find unseen water filled refuge within the flume net, tear the net or find a gap under the cod end net apron. The few blue crabs that attempted stranding were easily spotted. Smaller crabs were frequently observed climbing up the cod-end net mesh, though none were observed to successfully climb out.

Visual inspection of the cod-end and flume net during the 0% recovery trial revealed nothing abnormal and no obvious escape routes or net tears. A similar number of unmarked crabs and fish were captured in this *in situ* recovery efficiency trial as the other flume nets sampled during the same tide. These observations and the capture of large numbers of crabs from riprap throughout the study lead me to believe that blue crab recovery rates from riprap are higher than found during my two trials, and that the 0% recovery is an outlier. Due to the low recovery efficiency replication number, it is impossible to know if such escape events were restricted solely to riprap nets. Thus, it is possible that escape events may have occurred in both riprap and fringe marsh flume nets. Blue crab comparisons should be evaluated with the difference in recovery efficiency in mind. More blue crab recovery efficiency trials would have been desirable.

The presence of large nekton, including white mullet up to 120 mm TL, white perch up to 260 mm TL, and a 445 mm TL American eel suggest that even larger nekton were effectively trapped by the remotely tripped block nets. The high recovery efficiency results for all sizes of mummichogs in this study suggest that once fish were trapped by the block net there was little chance of escape from the flume net, even for the smallest nekton.

Relationship of Nekton Abundance and Intertidal Area Flooded

Hydroperiod has been shown in several studies to be the most important factor controlling nekton use of intertidal habitats (Rozas 1995, Yozzo and Smith 1998, Kneib 2000). Results from the 1st sampling period suggest however, that grass shrimp, mummichog and blue crab utilization of narrow intertidal habitats in Sarah Creek were not related to tide height or the amount of flooded intertidal area. Fitz and Wiegert (1991), Ayers (1995) and Minello and Webb (1997) also did not find an association between nekton abundance in the salt marsh and tidal height or tidal flooding distance into the marsh. Relationships between nekton abundance and the amount of intertidal area flooded may not have been apparent in this study for several reasons. Variation in tidal height and flood distance may not have been large enough for patterns to be observed, because block nets were always tripped at high tide during spring tides. A sampling design that sampled over a variety of tidal stages and amplitudes would be more appropriate to answer this question. Nekton abundance was also very patchy for the individual net results used for the analysis. Nekton distribution may also be restricted primarily to the edge of the intertidal areas sampled, as Cicchetti (1998a) observed for several species in the Goodwin Island marshes. The dynamics of nekton use in narrow intertidal habitats deserves more attention. Further work could elucidate the minimum fringe marsh width necessary to support local intertidal nekton populations.

Habitat Comparison: Total Abundance and Biomass per Meter.

Linear meter of shoreline results are more valid to compare habitat value of fringe marsh vs. riprap for several reasons. First, nekton density is generally greater on marsh edge compared with inner marsh (Kneib 2000, Zimmerman et al. 2000). Blue crabs, grass shrimp, Atlantic silverside, naked goby and all transient fish species were all found in greater numbers on the marsh edge compared to samples taken further into the marsh at the Goodwin Islands (Cicchetti 1998a). Abundance and biomass per square meter results for known edge species would be inaccurately decreased in the fringe marsh because total values for these species are divided by a greater intertidal area in the fringe marsh. Second, a positive relationship between grass shrimp, blue crab and mummichog abundance and intertidal area flooded was not observed in this study. This suggests that the habitat type is a bigger determinant of intertidal utilization than the amount of intertidal area flooded in the narrow intertidal areas sampled here. Third, the amount of edge was sampled evenly in both habitats. Fourth, the narrowest fringing marshes available were sampled in order to make the most valid possible comparison to flooded riprap. Mean flood distance in this study was 3.0 m in the fringe marsh and 1.9 m in the riprap. Fifth, installation of riprap occurs and is reported on a linear meter of shoreline basis (Barnard et al. 2001). In cases where fringe marsh exists, or could exist through site preparation, placement of riprap negates any future presence of fringe marsh on a linear meter of shoreline basis. Thus, riprap replaces fringe marsh on a linear meter of shoreline basis. Finally, results per linear meter of marsh have been used in several flume net studies to compare utilization of marsh habitats with no mention of differences in flooding between compared sites (McIvor and Odum 1986, Rozas et al. 1988, Hettler 1989). While the flume net sides were longer in these studies, often extending 'into the high marsh', there is no mention of how far the tide actually moved into, or beyond the flume nets, leaving no way to know if flume nets sampled similar area. Therefore, comparisons between these studies and this work can only be made on a per linear meter of edge basis. All of these points suggest that abundance and biomass per linear meter of shoreline results are a more useful parameter in order to compare nekton use of intertidal riprap and narrow fringing marsh.

Results per linear meter of shoreline reveal that the intertidal fringe marsh sampled here is a more heavily utilized habitat than the intertidal riprap. Abundance was significantly greater in the fringe marsh of all species with more than 25 individuals captured, (except grass shrimp and naked goby) including all fish species that are considered to be commercially/recreationally important. (Table 20). This indicates a negative impact to these species due to replacement of fringe marsh habitat with riprap. Abundance per linear meter of grass shrimp and naked goby was similar in both habitats, indicating no impact due to habitat replacement for these two species. The greater total number and relative abundance of small blue crabs, mummichog and naked gobies captured in the fringe marsh suggests that fringe marsh is a better nursery habitat for juveniles of these species (Figure 12 a,b,c). Species diversity was greater in the fringe marsh, and relative abundance was more evenly distributed in the fringe marsh (Figure 10), indicative of a more natural community (Clarke and Warwick 1994).

Biomass of total fish per linear meter of shoreline edge was significantly greater in the fringe marsh. Biomass per linear meter of shoreline edge of the three most dominant nekton species in the Sarah Creek intertidal (grass shrimp, blue crab, mummichog) was not different between marsh and riprap shorelines sampled here. The presence of larger mummichogs and blue crabs in the riprap increase biomass per linear meter values in the riprap to similar levels as the fringe marsh even though significantly more blue crabs and mummichogs were captured per linear meter of fringe marsh. Total biomass in riprap edge was 95.4% of that of fringe marsh edge, however total biomass was more evenly distributed among fish species in the fringe marsh (Figure 11). Biomass of striped killifish, Atlantic silverside and all commercially/recreationally important fish species was almost nonexistent in intertidal riprap (Table 12, Figure 9). Summer flounder, white perch and spot are valuable fishery species. These commercially important species and Atlantic silversides transport energy from productive shallow areas to deeper waters during their life cycle (Conover and Ross 1982). This trophic relay is an important component of the shallow water estuarine nekton community (Kneib 1997).

Habitat Comparison: Density and Biomass per Square Meter

A common assumption in nekton utilization studies is that a greater density of nekton in a habitat indicates better habitat value (Minello and Webb 1997, Able 1999). Density results in this study, however, could be misleading because flume nets sample two intertidal habitats simultaneously (edge and interior). Density per square meter values for species that preferentially utilize the edge of intertidal habitats will be 'diluted' to a greater extent in fringe marsh because total nekton were divided by total flood area which was approximately 58% greater in the fringe marsh (Hettler 1989). Density per square meter estimates for edge species in this study would show an apparent preference for the habitat with less intertidal area (riprap) when there really was no preference. Flume nets, as used in this study, do not have the resolution to elucidate species that preferentially utilize the edge of intertidal habitats.

Given the shortcomings of square meter results discussed above, density and biomass per square meter values indicate that riprap is utilized by marsh resident species except striped killifish. Several dominant intertidal nekton species in Sarah Creek were found in similar (e.g. mummichog density, Atlantic silverside density and biomass) or greater (e.g. mummichog biomass, grass shrimp density and biomass, naked goby density and biomass) extent per square meter in riprap. Blue crab density per square meter was significantly greater in the fringe marsh, however observed differences in blue crab recovery efficiency between riprap and fringe marsh make conclusions tenuous. Density and biomass per square meter of summer flounder, spot and white perch, all edge species, were significantly greater in the fringe marsh. These species displayed such a strong preference for fringe marsh habitat that results per square meter were still significantly greater in the fringe marsh even though fringe marsh values of these known edge species were divided by approximately 58% greater area. Density and biomass of striped killifish, a marsh resident species known to utilize all intertidal marsh habitat available, was also significantly greater in fringe marsh.

Species Discussion

Grass Shrimp

Grass shrimp are known to utilize all microhabitats of the flooded salt marsh, including pits and pools that remain filled at low tide (Yozzo and Smith 1998, Cicchetti 1998a). Peterson and Turner (1994) captured greater total numbers of *P. pugio* in flume nets that extended further into the flooded marsh (up to 40 meters), suggesting that total grass shrimp abundance in intertidal marsh increases with greater area of marsh flooded. Grass shrimp abundance, however, did not show a positive relationship with greater flooded area during the first sampling period, indicating that they move into fringing intertidal areas in an area-independent pattern. Abundance and biomass per linear meter results suggest that *P. pugio* in Sarah Creek is not impacted by the presence of intertidal riprap.

Grass shrimp, an important prey item for larger nekton, are attracted to structurally complex habitats and have a marked preference for vegetated habitats compared to unvegetated habitats (Kneib and Wagner 1994). Grass shrimp feed on detrital plant material, oligochaetes, nematodes, small benthic invertebrates and even small mummichogs (Welsh 1975, Anderson 1985, Kneib 1987). Sufficient food sources must be present in the intertidal riprap to support such high numbers of grass shrimp. Riprap, a spatially complex habitat, may also present lower rates of predation for grass shrimp than does fringe marsh, because fewer predators (striped killifish, summer flounder, white perch and spot) were found in the riprap. There was no difference in grass shrimp size between riprap and fringe marsh; further evidence that riprap habitat is suitable habitat for these animals.

Grass shrimp were captured in much greater density in both habitats in this study compared to studies performed on Goodwin Island (Table 21). Grass shrimp were observed on the flume net walls during high tide. Attraction to sampling gear would exaggerate quantitative density results, but comparison between habitats would still be valid for two reasons. First, attracted animals may have remained on the outside edge of the flume net wing, and would not then be captured. Second, attracted individuals could only be pulled from the immediate vicinity, thus results would represent the population of the proximal intertidal area. While population parameters may be exaggerated due to grass shrimp attraction to the flume net sides, it is clear that large numbers of grass shrimp utilize intertidal shorelines of Sarah Creek.

Blue Crab

Low replication of blue crab recovery efficiency makes it difficult to know if results reflect recovery efficiency, and the lower recovery efficiency in riprap makes comparison of blue crab population parameters difficult. Abundance per linear meter and density per square meter results indicate that blue crabs, known to be habitat opportunists, displayed a preference for fringing salt marsh habitat over riprap. Many studies report significantly greater number of blue crabs in salt marsh compared to shallow unvegetated areas (Zimmerman et al. 2000). Salt marshes are a documented nursery and foraging area for blue crabs (Zimmerman and Minello 1984, Fitz and Wiegert 1991). Blue crabs have a wide ranging diet that includes fish (including mummichogs), non-portunid crabs, plant matter and detritus, grass shrimp, small crustaceans, marsh periwinkles, softshell clams and each other (Kneib 1987, Hines and Ruiz 1995, Zimmerman et al. 2000). Larger crabs are safer from predation (Dittel et al. 1995), thus may be able to exploit a wider range of habitats than smaller individuals, which were more numerous in the fringe marsh.

Blue crabs were present in similar densities in both the fringe marsh and riprap when compared to Goodwin Island marsh edge (Table 21, Cicchetti 1998a). However, blue crab biomass per square meter was almost three times greater in the Sarah Creek intertidal than edge habitats at nearby Goodwin Island (Table 22). Such comparison makes it clear that narrow intertidal areas of Sarah Creek are utilized by blue crabs.

Mummichog

Mummichog are marsh resident fish that use all available flooded marsh habitats, including water filled pits, pools and depressions on the marsh surface during low tide. Cicchetti (1998a) found significantly greater mummichog density in interior marsh habitats compared to marsh edge at the Goodwin Islands. However, mummichog abundance did not increase with greater tidal heights during the first sample period as would be expected for species that utilize interior marsh habitats. Almost two times as many mummichog were captured in the fringe marsh during this study. Abundance per linear meter was significantly greater in the fringe marsh compared to the riprap, indicating that riprap negatively impacts mummichog.

Mummichog larvae (TL < 25mm) comprised similar proportions of the total mummichog abundance in both habitats, although more than twice as many larval sized mummichogs were found in the fringe marsh (Figure 12b, Figure 13c). Median length was consistently smaller in fringe marsh, and large mummichogs (> 60 mm) comprised a larger percentage of the mummichog population in riprap, indicating that fringe marsh is a better habitat for small mummichogs (Table 24a, Figure 12b). Intertidal salt marsh is a known nursery for larval mummichogs (Kneib and Wagner 1994). Mummichogs were found in greater densities in narrow intertidal habitats in Sarah Creek than in the much larger marshes of Goodwin Islands, VA (Table 21).

Mummichogs rely on vision and possibly olfaction, when foraging and they feed predominately during the day (Zimmerman et al. 2000, Abraham 1985). Prey items include small crustaceans (copepods, amphipods, isopods, tanaids, fiddler crabs and grass shrimp), polychaetes, insects, ostracods, detritus, algae, fish and fish eggs (Kneib and Stiven 1982, Abraham 1985, Kneib 1987). Access to intertidal marshes has been shown to be necessary for normal mummichog growth (Weisberg and Lotrich 1982). The presence of similar mummichog density in the riprap suggests that food sources do exist for mummichog in the intertidal riprap.

Naked Goby

Previous marsh utilization studies have found naked gobies mainly in the marsh edge, closest to open water (Minello and Webb 1997, Peterson and Turner 1994). Statistical comparison of linear meter of edge results suggest that naked gobies utilize both habitats to a similar degree. However, larger naked goby were found in the riprap. Thus, naked goby biomass per linear meter was greater per meter of riprap edge.

Naked goby, known for a close association with oyster reef and SAV habitats in the Chesapeake Bay, comprised only 0.6% of the total fish captured in flume weirs on the Goodwin Island marshes in 1994 (Ayers 1995) and 0.03% of the total fish captured in the marsh with drop rings on Goodwin Island in 1995 (Cicchetti 1998a). Naked gobies, known to exhibit behavioral flexibility, were the second most abundant fish in this study, the third most abundant fish captured in pit traps on Eastern Shore marshes (Yozzo and Smith 1998), the twelfth most abundant fish on North Carolina marshes (Hettler 1989) and the most abundant fish on Gulf of Mexico marshes (Zimmerman and Minello 1984, Peterson and Turner 1994). Use of the intertidal marsh may be related to presence or absence of local SAV or oyster reef habitats. Lack of both of these in Sarah Creek may increase the relative importance of fringing intertidal habitats to this species.

Altantic silversides

Atlantic silversides were found to utilize mainly the marsh edge at the Goodwin Islands (Cicchetti 1998a), therefore results reported as abundance per linear meter are a better indication of habitat selection. Atlantic silversides displayed a preference for intertidal fringe marsh in this study, as more than twice as many were captured in fringe marsh compared to the riprap. Almost half of the total silversides (21 individuals) that were captured in the riprap were found in one net. Silverside catch was more evenly distributed in fringe marsh samples.

The majority (76/101) of silversides captured in the fringe marsh were caught at night. Cicchetti (1998b) also observed significantly greater marsh use at night. Silversides feed on mysids, copepods and other small crustaceans during the daytime (Murdy et al. 1997). The majority of individuals captured in Goodwin Island on the marsh at night had mostly empty stomachs (Cicchetti 1998b). Cicchetti (1998b) proposed that Atlantic silversides use the marsh at night as a predation refuge in order to avoid nocturnal predators that move into shallow waters during night time high tides (Austin 1996).

Summer Flounder

The vast majority of summer flounder were captured in fringe marsh at night during the May 4-8 sampling period. Although only one site was sampled during this time, it is apparent that juvenile summer flounder have a strong preference for intertidal fringe marsh habitat over riprap. Juvenile summer flounder use intertidal unvegetated areas along the York River from February to early June. Intertidal abundance peaks in the intertidal zone during April, and juveniles leave the shallow intertidal for the most part by June (CBNERRVA 2001). Ayers (1995) and Hettler (1989) report capturing summer flounder on the marsh surface, although in less abundance than found during the May 4-8 sampling period in this study. Temporal trends in marsh utilization were not apparent during either study and diurnal sampling was not conducted. Cicchetti (1998a) did not catch any summer flounder on the Goodwin Island marshes during his habitat use study that began in June, or during a diel study that took place in August and September, well after young of the year have moved to deeper habitats.

Striped Killifish

Striped killifish exhibited a strong preference for marsh habitat. Striped killifish utilize all marsh habitats, with greater abundance found further from the edge. They move off the marsh as the tide ebbs and use shallow (< 10 cm deep) unvegetated habitat during low tide (Cicchetti 1998a). The majority of striped killifish captured by Ayers (1995) were caught on marshes with a more protected shoreline. This species is normally less dominant in terms of abundance compared to mummichogs in most marsh habitats. They were the third most abundant fish captured on the Goodwin Island marsh surface in 1994, comprising 2.6% of sampled fish (Ayers 1995). Striped killifish density on the fringe marsh was 0.32 inds $\cdot$ m^{-2} in this study. This number is comparable to striped killifish density at Goodwin Island in 1996, which averaged 0.16 inds $\cdot$ m^{-2} on the marsh edge, 0.11 inds $\cdot$ m^{-2} on the marsh fringe and 0.47 inds $\cdot$ m^{-2} on interior marshes (Cicchetti 1998a).

Striped killifish have are omnivorous, similar to mummichog. Grass shrimp, algae, fish (including cannibalism), insect larvae, polychaetes, copepods, amphipods and other small crustaceans are known to be consumed (Abraham 1985). Striped killifish are known to rely more heavily on benthic invertebrates while mummichogs consume more epiphytic algae. Striped killifish spawn in protected shallow water close to shore and in marsh ponds from April through September, and they are thought to bury their eggs (Abraham 1985). More information is needed in order to surmise whether forage base, spawning area, nursery area or predation play a role in the absence of striped killifish from the riprap.

White Perch and Spot

White perch and spot displayed a strong preference for fringe marsh in this study. White perch were not captured on the marsh or marsh edge in previous marsh utilization studies at the Goodwin Islands, which are located downriver from Sarah Creek (Ayers 1995, Cicchetti 1998a and c). White perch are ubiquitous in fresh and brackish waters of the Chesapeake Bay (Murdy et al. 1997). Smaller individuals feed on mysids and amphipods, and larger individuals feed on clam siphons, crabs, shrimp and small fish (Smith et al. 1984, Murdy et al.1997).

Spot density averaged 0.1 inds $\cdot$ m^{-2} in the fringe marsh. While this is well below the density (1.2 inds $\cdot$ m^{-2}) reported by Ayers (1995), Ayers reports density of the species only for the time period they were captured, not for the duration of the study as calculated here. Cicchetti reports catching 0.05 inds $\cdot$ m^{-2} on the marsh edge, a number similar to what was found in this study. Varnell and Havens (1995) report total sciaenid mean density of 0.01 sciaenids $\cdot$ m^{-2} in the summer to 0.27 sciaenids $\cdot$ m^{-2} in the spring in two intertidal creek areas near the headwaters of Sarah Creek. The majority (96%) of these sciaenids were spot. Both spot and white perch (family: Moronidae) density averaged 0.1 inds $\cdot$ m^{-2} in this study, the cumulative total of which is comparable to density of sciaenids found by Varnell and Havens (1995).

Spot, a known marsh transient, are found in much greater abundance in marshes in the Southern United States. They were the second most abundant fish captured on marshes in North Carolina (Hettler 1989). Polychaetes, nematodes, maldanids and harpactacoid copepods are significant food sources of spot, which are known to forage in the shallows (Smith et al. 1984, Miltner et al. 1994). Spot feed predominately during the day (Middleton 1986). The benthic invertebrates that spot feed on are known to be more abundant in or near Spartina marshes than unvegetated areas (Miltner et al. 1994). Smaller juveniles select for shallow, muddy substrate areas while larger juveniles move to deeper estuarine waters (Hales and Van Den Avyle 1989). Spot were the dominant fish species in trawl studies in tidal creeks of the York River (Smith et al. 1984).

Marsh Utilization: Comparison with Other Studies

Sixteen fish species were captured on the marsh surface in this study. This number is comparable to the 13 fish species captured by Cicchetti (1998b) and 11 species captured by Ayers at the Goodwin Islands. No *Cyprinodon variegatus* (sheepshead minnow) and only two rainwater killifish and were captured in this study. These two marsh resident species were captured in significant abundance in the two Goodwin Island studies. Diversity in the mesohaline marshes sampled here is low compared to marshes in different locations. McIvor and Odum (1986) captured 21 fish species in tidal freshwater marshes on the James River, VA. Hettler (1989) captured 35 species of fish from flume nets in N. Carolina and Peterson and Turner (1994) captured 29 species of fish on Louisiana salt marshes.

The narrow fringe marshes sampled in Sarah Creek support similar, if not greater nekton density and biomass per square meter than larger marshes at the Goodwin Islands, highlighting the importance of intertidal habitats in creek systems (Table 21 and 22). Similarly, total abundance and biomass per meter of marsh edge for mummichog, naked goby, Atlantic silverside and blue crabs in Sarah Creek were comparable to, or greater than, values found on North Carolina marshes that had 5 to 11 times greater flooded marsh area (Table 23). Total fish abundance and biomass was greater in the North Carolina study due to heavy marsh utilization by spot, striped killifish, sheepshead minnow, spotfin mojarra *(Eucinostomus argeneus)*, pinfish *(Lagodon rhomboides),* and white mullet (Hettler 1989). While direct comparison can not be made due to temporal and physical site variation and sample gear differences, it is clear that narrow fringe marshes are a highly utilized habitat. Behavioral flexibility of estuarine species such as blue crabs, mummichog and grass shrimp may increase the relative importance of seemingly marginal marshes in areas like Sarah Creek where preferred habitat, such as oyster reef and SAV, have disappeared.

Peak blue crab abundance during an erosional marsh edge study at the Goodwin Islands was 1.74 crabs per meter of edge (Cicchetti 1998c). This number is low compared to the 10.9 blue crabs per meter of fringe marsh in this study, most likely because the marsh surface was not sampled in the erosional edge study. Commercially important fish species abundance was between 1.25 individuals per meter of erosional marsh edge during the period of peak abundance (93% spot) and 0.6 individuals per meter during the period of peak fishery species diversity (37% spot, Cicchetti 1998c). These values are similar to commercially important fishery species abundance per meter of fringe marsh edge in this study, which averaged 1.8 (SE = 0.6) individuals per meter of fringe marsh edge over the duration of the study. The peak of commercially important fish abundance during the May 4-8 sample period (5.4 inds. per meter of fringe marsh edge, SE = 1.7) is greater than the abundance of any fishery species found in either depositional or erosional marsh edge on the Goodwin Islands (Ayers 1995, Cicchetti 1998a and c).

The high nekton abundance in the narrow marshes sampled here support the importance of vegetated edge habitat and tidal creeks (Minello et al. 1994, Baltz et al. 1993, Cicchetti 1998a, Cicchetti and Diaz 2000, West and Zedler 2000). Total fish density is 3.5 times greater and blue crab density is 2 orders of magnitude greater in the marsh edge habitats sampled in this study compared to total fish and blue crab density found over an entire intertidal marsh creek system at the headwaters of Sarah Creek (Varnell and Havens 1995). Direct comparison with this study is difficult because total fyke net catch was divided by total intertidal marsh and creek area and animals may have escaped capture by residing in small water filled depressions at low tide. However, the much greater density of fish and crabs captured in the intertidal edges in this study compared to the whole intertidal creek system discussed above suggest that nekton abundance is focussed along edge habitat.

Summary and Management Implications

This study is the first time, to my knowledge, that nekton has been quantitatively sampled from intertidal revetment habitat. Almost two times as many fish were captured per linear meter of fringe marsh shoreline. Nekton standing stocks are often significantly greater in vegetated compared to unvegetated aquatic habitats (Zimmerman and Minello 1984, Kneib and Wagner 1994). Although the difference between abundance in vegetatated vs. unvegetated habitat is not as disparate as other studies, riprap structure did not attract any species, in significant numbers, that were not also found in the fringe marsh. Therefore intertidal riprap does not share the reef like attributes observed in erosion control rock structures in non-tidal fresh water shorelines and subtidal jetties (Hay and Sutherland 1988, Beauchamp et al. 1994). Instead, riprap structure seemed to exclude species that were captured in large numbers in the fringe marsh including striped killifish (marsh resident), white perch (estuarine resident) and summer flounder and spot (both estuarine transients). All of the commercially and recreationally important fishery species (possibly including blue crabs) show a preference for fringe marsh. Total nekton abundance and biomass is distributed among fewer species in the riprap, indicating a less robust nekton community utilizing riprap compared to fringing marsh. Fringing marsh also supports a more diverse nekton community.

Total abundance of seven of the nine numerically dominant species captured in this study were all significantly greater per meter of fringe marsh shoreline (grass shrimp and naked gobies were captured in statistically similar abundance per linear meter in either habitat). Biomass of fishery species (summer flounder, white perch, spot, white mullet, silver perch, seatrout and spadefish), and striped killifish and silversides (both ecologically important species), was virtually nonexistent in intertidal riprap. Thus, trophic transfer of intertidal productivity to deeper water is compromised in shorelines with intertidal riprap. Density estimates show that riprap habitat was utilized by mummichog, grass shrimp, blue crab, naked goby and Atlantic silversides, the dominant nekton found in this study. Biomass of the dominant estuarine resident nekton in Sarah Creek is not diminished in intertidal riprap, however, the presence of a greater number of small mummichog and blue crabs in the fringe marsh suggests that fringe marsh is a better nursery area for these species.

The nature and prevalence of edge habitats is an important facet in estuarine ecosystems (Nordstrom and Roman 1996, Cicchetti 1998b). In light of increased shoreline development and rising sea levels, the demand for shoreline hardening will at least remain at present levels. Cumulative habitat degradation and coastal habitat loss through physical alteration and other anthropogenic activities have had adverse effects of fish populations (Sarthou 1999). The installation of shoreline hardening structures is largely irreversible. It is clear that the narrow fringing marshes are a better overall habitat than the intertidal riprap sampled in Sarah Creek. Managers should therefore work to maintain existing fringe marsh. According to the Virginia Marine Resources Commission (1993), marsh grass planting is the preferred method to stabilize banks along shorelines with low fetch distances (< 0.5 nautical miles according to Barnard 1994, < 1.0 nautical mile according to Hardaway and Byrne 1995). However, marsh planting projects are very rarely observed and riprap is commonly installed along low energy shorelines such as Sarah Creek. Fringing marshes should be preserved, by using offshore breakwaters or marsh toe revetments or restored, through various site preparation techniques wherever possible. In instances when riprap is the only possible alternative, the efficacy of riprap with a more gradual slope should be investigated. Development of alternative shoreline hardening methods that allow exchange between the natural intertidal areas and the subtidal should be also be explored. The current permitting process looks favorably on proposals that place riprap high enough on the bank to allow fringing marsh to grow in front of the rock structure. While this preserves highly utilized fringe marsh edge habitat, thought must be given to the future of such fringe marsh in light of apparent sea level rise.

Future Investigation

The intertidal shoreline areas sampled were located within a mosaic of natural and hardened shoreline. It is possible that such a mixed landscape served to blur results, as sampled riprap was never far from *Spartina* marsh. Future work should attempt to sample from larger and more isolated stretches of intertidal habitat in order to keep landscape ecology issues from potentially masking patterns. Research into the nekton populations in small subestuaries with a continuum of shoreline hardening may provide evidence that impacts observed to the intertidal nekton community along riprap intertidal areas in this study may effect larger scale nekton community structure. Studies comparing shorelines not sampled here, such as eroding marsh, eroding unvegetated bank, fringe marsh in front of riprap and sandy beach would also provide valuable information. Future investigation into nekton growth rate, survival rate and input into adult populations and silverside and fundulid spawning in intertidal riprap and narrow fringe marsh would also be useful. Investigation into the microhabitat use of fringe marshes (e.g. edge vs. interior, critical fringe width) would also give managers important information for future permitting policy and marsh restoration projects.

Table 1. Species Composition of Fringe Marshes.

Site A	Site B	Site C
Spartina alterniflora – Saltmarsh cordgrass	*Spartina alterniflora*	*Spartina alterniflora*
Scirpus robustus – Saltmarsh bulrush threesquare		*Juncus roemerianus* – Black needlerush
Typha angustifolia – Narrow-leaved cattail		*Scirpus robustus*
Aster tenuifolius – Saltmarsh aster		*Aster tenuifolius*
Spartina patens – Saltmeadow hay		*Spartina patens*
Distichlis spicata - Salt grass		**Atriplex patula* – Spearscale
**Baccaris halimifolia* – Groundsel tree		**Iva frutescens*
**Iva frutescens* – Marsh elder		**Baccaris halimifolia*
		**Panicum virginicum* – Switch grass

* denotes plant species which were flooded only at the highest tides.

Parameter	Fringing Marsh Site A	Fringing Marsh Site B	Fringing Marsh Site C
# of Flume Net Stations	3	2	2
Length of Shoreline	72 m	48 m	70 m
Adjacent Shoreline	Fringe marsh	Bulkhead Pocket marsh	Fringe marsh
Offshore Slope	6.0%	4.3%	4.7%
Marsh Slope	16.2%	16.3%	20.3%
Sediment	Gravel: 0.4% Sand: 79.9% Silt: 7.7% Clay: 12.0%	Gravel: 0.0% Sand: 89.4% Silt: 5.9% Clay: 4.7%	Gravel: 0.2% Sand: 91.3% Silt: 2.7% Clay: 5.8%
Avg. Marsh Width N = 10	3.4 m Std Dev = 0.5	2.7 m Std Dev = 0.5	3.9 m Std Dev = 0.5
Avg. Stem Density N = 10	$42.8 \cdot 0.25\ m^{-2}$ Std Dev = 26.7	$66.8 \cdot 0.25m^{-2}$ Std Dev = 14.7	$63.0 \cdot 0.25m^{-2}$ Std Dev = 23.6
Marsh Flora	Mixed Mesohaline*	*Spartina alterniflora*	Mixed Mesohaline*
Characteristics Behind Marsh	0.5 – 0.8 m high bank	Isolated marsh pond	1.2 – 1.5 m high bank

* See Table 1 for species composition of mesohaline plants found at the these study cites

Table 2. Site Parameters of Fringing Marsh Sites, Sarah Creek, VA.

Length of shoreline habitat is the length of shoreline from which the flume net stations were randomly selected. Marsh slope denotes the average slope of the marsh regularly flooded during spring high tides. Offshore slope denotes the average slope from the marsh or riprap edge to approximately 15 meters offshore.

Parameter	Revetment Site Description A	B	C
# of Flume Net Stations	3	2	2
Length of Structure	68 m	20 m	60 m
Adjacent Shoreline	Wood Bulkhead Pocket marsh	Concrete Bulkhead Riprap w/ fringe marsh	Fringing marsh Riprap
Distance to paired fringe marsh site.	50 m	200 m	500 m
Offshore slope	6.4%	3.4%	4.3%
Revetment sub-surface slope	34.5%	22.4%	34.2%
Rock surface slope	56.0%	37.8%	43.3%
Sediment	Gravel: 0.1% Sand: 88.8% Silt: 4.8% Clay: 6.2%	Gravel: 0.3% Sand: 93.1% Silt: 0.2% Clay: 6.5%	Gravel: 1.1% Sand: 87.6% Silt: 4.3% Clay: 7.1%

Table 3. Site Parameters of the Intertidal Revetment Sites, Sarah Creek, VA.

Length of revetment is the length of shoreline from which flume net stations were randomly selected. Revetment subsurface slope denotes the slope of the ground under the riprap rocks regularly flooded during spring high tides. Rocks were removed down to the filter cloth to measure subsurface slope. Rock surface slope denotes the slope of the portion of the riprap rock surface regularly flooded during spring high tides. Offshore slope denotes the average slope from the revetment edge to approximately 15 meters offshore.

Table 4. Sampling Schedule.

Number of high tides sampled at each site during each sampling period (bold numbers). Total number of individual marsh and riprap flume net samples taken during that period listed (regular font). Moon phase at each sampling period is shown.

Table 4. Sampling Schedule.

Sampling Period	1	2	3	4	5	6	Total Tides Sampled
Dates	5/4 – 5/8	5/16 – 5/18	6/1 – 6/6	6/28 – 7/05	8/1 – 8/2	8/27 – 9/02	
Moon phase	New - 5/3	Full – 5/18	New – 6/2	New – 7/1	New – 7/30	New – 8/28	
A **Marsh** (# of Nets)	17	6	5	6	Not Sampled	4	38
A (# of Tides)	**6**	**2**	**2**	**2**		**2**	**14**
A **Riprap** (# of Nets)	17	6	4	5		4	36
B **Marsh**	Not Sampled	6	4	2	Not Sampled	4	16
B (# of Tides)		**3**	**2**	**1**		**2**	**8**
B **Riprap**		6	4	2		4	16
C **Marsh**	Not Sampled	Not Sampled	3	7	5	2	17
C (# of Tides)			**2**	**4**	**3**	**1**	**10**
C **Riprap**			2	8	6	2	18

Grand Total N = 32 Tides Sampled
n = 141 Flume Nets (Marsh = 71, Riprap = 70)

Table 5. Length-weight regressions used for biomass analysis.

Table 5. Length-Weight Regressions Used for Biomass Analysis.

Species	Size Range (mm)	Measurement (mm)	n	Equation	R^2	Source
Callinectes sapidus	5-30	point to point (CW)	42	gww = 0.00008794 * CW ^ 3.0201	0.93	Cicchetti 1998a
Callinectes sapidus	33-119	point to point	115	gww = 0.0001152 * CW ^ 2.9338	0.95	Cicchetti 1998a
Callinectes sapidus	> 120	point to point	1,242	gww = 0.00062420 * CW ^ 2.55	0.92	Olmi and Bishop 1983
Fundulus heteroclitus	7 - 20	total length (TL)	70	gww = 0.000009856 * TL ^ 2.9634	0.73	Cicchetti 1998a
Fundulus heteroclitus	21-100	total length	452	gww = 0.000004796 * TL ^ 3.2718	0.97	Cicchetti 1998a
Gobiosoma bosc	9-19	total length	28	gww = 0.000007826 * TL ^ 3.0951	0.88	Cicchetti 1998a
Gobiosoma bosc	20-52	total length	94	gww = 0.000005168 * TL ^ 3.2397	0.96	Cicchetti 1998a
Menidia menidia	12-21	fork length (FL)	26	gww = 0.00000007046 * FL ^ 4.5147	0.75	Cicchetti 1998a
Menidia menidia	22 - 92	fork length	196	gww = 0.00001455 * FL ^ 2.8501	0.95	Cicchetti 1998a
Paralichthys dentatus	42 - 78	total length	30	gww = 0.0772 * e ^ 0.0525 * TL	0.98	This study
Fundulus majalis	24 - 111	total length	87	gww = 0.000004453 * TL ^ 3.2385	0.98	Cicchetti 1998a
Morone americana	28 - 54	total length	16	gww = 0.0157 * e ^ 0.0915 * TL	0.93	This study
Leiostomus xanthurus	42 - 73	total length	10	gww = 0.0904 * e ^ 0.0545 * TL	0.95	This study
Palaemonetes pugio	12-42	Total length (rostrum – telson)	204	gww = 0.00001030 * TL ^ 2.9775	0.98	Cicchetti 1998a

gww = grams wet weight

Table 6. Statistical Results of Physical Site Parameters.

Results of paired t-tests on physical site parameters. Temperature, turbidity, salinity and dissolved oxygen data were successfully recorded at both marsh and riprap sample areas on 20 of the sampled tides. Tidal heights at the mouth of the flume net were recorded for every flume net immediately after block nets were tripped. Area flooded was flagged and measured for every marsh flume net. Area flooded in the riprap was calculated for tide heights during installation of flume nets.

Parameter	Fringe Marsh Value	Riprap Value	DF	Test Result t	P Value
Temperature *	24.6 C	24.2 C	19	2.41	0.026
Turbidity *	17.0 NTU	15.0 NTU	19	2.73	0.013
Salinity	17.6	17.6	19	0.22	0.829
Dissolved Oxygen	6.7 mg/L	6.7 mg/L	18	0.48	0.635
Tidal Height *	57 cm	59 cm	30	-2.78	0.009
Area Flooded *	3.0 m^{-2}	1.9 m^{-2}	30	9.68	< 0.001

* denotes significance at the $\alpha = 0.05$ level

Table 7. Mean Tidal Height and Flooded Intertidal Area by Site.

The mean tidal height (cm) at the mouth of the flume nets at the time the block nets were tripped is given for each site. Average tide height and area flooded of fringe marsh and riprap flume nets for each sample tide were used for the calculation (N site A = 14, B = 8, C = 9). Standard error and the range of mean tidal heights for sampled tides are given for each site. Mean intertidal area (m^{-2}) flooded at the time the block nets were tripped is also given with standard error and range. Results of one-way analysis of variance between fringe marsh and riprap sites are given. Results of Tukey's multiple comparison test are included when significance was found with ANOVA. Bars (over or under) site abbreviations denote non-significant differences between sites. No difference was observed in tidal height between the three fringe marsh sites or the three riprap sites.

Site	**Shoreline Type**	**Tidal Height (cm) and SE**	**Range (cm)**	**Flooded Area (m^{-2}) and SE**	**Range (m^{-2})**
A	Fringe Marsh	56 (2.7)	36 - 76	3.3 (0.2)	2.1 – 3.7
	Riprap	58 (2.5)	37 – 73	1.8 (0.1)	1.4 – 2.0
B	Fringe Marsh	53 (2.52)	42 – 60	2.7 (< 0.1)	2.6 – 2.7
	Riprap	59 (2.53)	48 – 68	2.4 (0.1)	2.2 – 2.4
	Fringe Marsh	61 (3.6)	40 – 72	2.9 (0.2)	1.7 – 3.3
C	Riprap	61 (3.4)	43 – 72	1.7 (< 0.1)	1.6 – 1.8

Tukey's results   Fringe Marsh Flooded Area   B C A

Riprap Flooded Area   C A B

Table 8. Flume Net Recovery Efficiencies

Flume net recovery efficiencies for *Fundulus heteroclitus* (35 – 90 mm TL), *Callinectes sapidus* (25 – 60 mm carapace width) and *Paralichthys dentatus* were estimated by placing marked individuals into the flume nets after the cod end nets were installed during data collection. Two different methods were used to estimate recovery efficiencies for mummichogs between 10 – 25 mm TL. Fish were bathed in Rose Bengal and released into the flumes on the stain trial. In the unmarked - recapture trials, nekton were kept out of the flume net enclosures and unmarked fish were released into the flume nets.

Table 8. Flume Net Recovery Efficiencies

Species	Size	Site	Shoreline	Trials	Inds.	Estimate	95% C.I.
Fundulus heteroclitus	35 - 90 mm TL	A	N	9	76	80%	61 – 99%
"	35 - 90 mm TL	A	RR	7	51	87%	74 – 100%
"	35 - 90 mm TL	B	N	4	30	100%	
"	35 - 90 mm TL	B	RR	2	10	100%	
"	35 - 90 mm TL	C	N	3	31	97%	89 – 100%
"	35 - 90 mm TL	C	RR	3	28	86%	79 – 93%
"	35 - 90 mm TL	**Total**	**N**	16	137	**88%**	**77 – 99%**
"	35 – 90 mm TL	**Total**	**RR**	12	89	**89%**	**81 – 97%**
Fundulus heteroclitus	15 – 25 mm TL	C	N	1	23	78%	* Stain trial
"	12 – 25 mm TL	C	RR	1	23	83%	* Stain trial
Fundulus heteroclitus	10 – 25 mm TL	C	N	2	100	95%	* Unmark-recap
"	10 – 25 mm TL	C	RR	2	100	68%	* Unmark–recap
Callinectes sapidus	25 – 60 mm CW	A	N	2	18	83%	
"	25 – 60 mm CW	A	RR	2	20	38%	
Paralichthys dentatus	35 – 60 mm TL	B	N	1	4	100%	

Table 9. Flume Net Recovery Efficiency Among Fringe Marsh and Riprap Sites.

ANOVA results of *Fundulus heteroclitus* recovery efficiency between the three fringe marsh sites.

Source	**DF**	**SS**	**MS**	**F**	**P**
Site	2	1391.75	695.9	1.721	0.217
Error	13	5255.60	404.3		
Total	15	6647.35			

ANOVA results of *F. heteroclitus* recovery efficiency between the three riprap sites.

Source	**DF**	**SS**	**MS**	**F**	**P**
Site	2	311.95	156.0	0.863	0.458
Error	8	1446.2	180.7		
Total	10				

Table 10. Statistical Results of Recovery Efficiency Between Habitats and Day/Night.

Students t-test results of *F. heteroclitus* recovery efficiency between fringe marsh and riprap flume net samples.

Shoreline	**N**	**Mean**	**Std. Dev**	**T**	**P**
Natural	16	88.1	21.1	-0.13	0.91
Rip-rap	12	88.9	12.7		
Total	DF = 25				

Students t-test results of F. heteroclitus recovery efficiency between day and night flume net samples.

Shoreline	**N**	**Mean**	**Std. Dev**	**T**	**P**
Day	8	88.8	13.3	-1.10	0.29
Night	8	77.2	26.7		
Total	DF = 14				

Table 11. Total Nekton Abundance, Relative Abundance and Index of Relative Abundance.

The total number of individuals captured in fringe marsh and riprap flume net samples is shown in column 1. Totals are calculated from 71 fringe marsh flume nets and 70 riprap flume nets. Relative abundance (total number of a fish species captured / total number of fish captured) of fish species in fringe marsh and riprap is shown in column 2. Percent occurrence (number of samples in which a fish species was captured / total number of samples) of fish species in fringe marsh and riprap is shown in column 3. The Index of Relative Importance (IRI), is shown in column 4. IRI is estimated as (relative abundance) * (percent occurrence). The IRI is used to minimize the importance of fish that are captured infrequently but in large numbers.

	Fringe Marsh				Riprap			
Species	**Total Inds.**	**% Abund.**	**% Occurr.**	**IRI**	**Total Inds.**	**% Abund.**	**% Occurr.**	**IRI**
Fundulus heteroclitus	945	66.0	95.8	6,322.8	514	64.2	98.6	6,330.1
Gobiosoma bosc	127	8.9	23.9	212.7	206	25.7	45.7	1,174.5
Menidia menidia	101	7.1	32.3	229.3	43	5.4	5.7	30.8
Paralichthys dentatus	91	6.4	19.7	126.1	7	0.9	7.1	6.4
Fundulus majalis	80	5.6	32.4	181.4	2	0.3	2.9	0.9
Morone americana	30	2.1	15.5	32.6	2	0.3	2.9	0.9
Leiostomus xanthurus	25	1.7	12.7	21.6	1	0.1	1.4	0.1
Symphurus plagiusa	11	0.8	8.5	6.8	0	0	0	0
Mugil curema	6	0.4	7.0	2.8	0	0	0	0
Morone saxatilis	2	0.1	2.8	0.3	0	0	0	0
Luciana parva	2	0.1	2.8	0.3	0	0	0	0
Bairdiella chrysoura	2	0.1	2.8	0.3	0	0	0	0
Cynoscion sp.	1	0.1	1.4	0.1	2	0.3	1.4	0.4
Pomatomus saltatrix	1	0.1	1.4	0.1	0	0	0	0
Chaetodipterus faber	1	0.1	1.4	0.1	0	0	0	0
Anguilla rostrata	0	0	0	0	1	0.1	1.4	0.1
Clupeid sp.	1	0.1	1.4	0.1	0	0	0	0
Chasmodes bosquianus	0	0	0	0	2	0.3	2.9	0.9
Gobiesox strumosus	0	0	0	0	8	1.0	7.1	7.1
Unidentified larvae	7	0.5			13	1.6		
Total Fish	1,432	100%		7,137.4	801	100%		7,552.4
Total Fish Species - 19	16				11			
Callinectes sapidus	765				385			
Palaemonetes pugio	14,354				16,414			
Penaed Shrimp	0				1			

Species	Marsh GDW	Marsh % of Total Fish GDW	Riprap GDW	Riprap % of Total Fish GDW
Mummichog, *Fundulus heteroclitus*	600.8	74.2	493.2	90.8
Naked goby, *Gobiosoma bosc*	12.8	1.6	36.8	6.8
Atlantic silverside, *Menidia menidia*	15.5	1.9	5.7	1.1
Summer flounder, *P. dentatus*	32.1	4.0	1.8	0.3
Striped killifish, *Fundulus majalis*	87.5	10.8	< 0.1	< 0.1
White perch, *Morone americana*	7.5	0.9	0.4	0.1
Spot, *Leiostomus xanthurus*	24.1	3.0	< 0.1	< 0.1
Blackcheek tonguefish, *S. plagiusa*	4.7	0.6	0	0
White mullet, *Mugil curema*	10.8	1.3	0	0
Striped bass, *Morone saxatilis*	0.7	0.1	0	0
Rainwater killifish, *Luciana parva*	< 0.1	< 0.1	0	0
Silver perch, *Bairdiella chrysoura*	7.4	0.9	0	0
Seatrout, *Cynoscion* sp.	4.6	0.6	0.3	0.1
Bluefish, *Pomatomus saltatrix*	0.2	< 0.1	0	0
Spadefish, *Chaetodipterus faber*	0.8	0.1	0	0
Striped blenny, *C. bosquianus*	0	0	2.7	0.5
Skilletfish, *Gobiesox strumosus*	0	0	2.4	0.4
Total Fish	**809.5**		**543.3**	
Blue crab, *Callinectes sapidus*	**1,634.2**		**1,587.9**	
Grass shrimp, *Palaemonetes pugio*	**1,259.8**		**1,403.1**	
Total Nekton Biomass	**3,703.5**		**3,534.3**	

Table 12. Total and Relative Biomass.

Total biomass (grams dry weight) of nekton species collected is given. Relative biomass (Total biomass of a fish species / Total biomass of all fish species in fringe marsh or riprap) is broken down for individual fish species. Totals are calculated from 71 fringe marsh flume net samples and 70 intertidal revetment flume net samples. Two large white perch (137.55 gdw total) were not included in fringe marsh biomass numbers. One large white perch (62.46 gdw) and one large American eel (47.66 gdw) were not included in riprap numbers.

Table 13. Abundance and Biomass of Nekton by Sampling Period and Overall Mean.

The number of individuals captured per meter of shoreline (inds m^{-1}) and per square meter (inds m^{-2}) and grams dry weight (gdw) biomass values per meter of shoreline (gdw m^{-1}) and per square meter (gdw m^{-2})are listed for total fish, *Callinectes sapidus*, *Palaemonetes pugio*, *Fundulus heteroclitus*, *Gobiosoma bosc*, *Menidia menidia*, *Paralichthys dentatus* and *Fundulus majalis*. Period means are grand means of the sites sampled during that period (Figure 4). For sampling periods 1 and 5, standard error depicts variation between sample tides during the period because only 1 site was sampled. Standard error depicts variation between sites for the other periods. The overall mean is the mean of the six sampling periods. These values are only used in text to describe temporal variations in nekton abundance and biomass during the study.

Species	Habitat	Data	Sampling Period Grand Means						Overall Mean
			5/4-8	5/16-18	6/1-6	6/28-7/5	8/1-2	8/27-9/2	
Total Fish	Marsh	inds m^{-1}	14.5	14.6	17.4	22.2	19.3	30.9	19.8
		SE	3.8	6.6	3.2	7.2	8.3	9.5	
		gdw m^{-1}	8.11	9.85	11.25	16.36	9.72	15.01	11.72
		SE	3.11	2.31	0.47	4.68	2.02	4.53	
		inds m^{-2}	4.9	5.5	5.3	7.2	6.0	9.2	6.4
		SE	1.3	2.0	0.7	1.5	2.7	2.4	
		gdw m^{-2}	2.45	3.47	3.5	5.07	3.06	4.3	3.64
		SE	0.87	0.4	0.07	1.04	0.69	0.92	
	Riprap	inds m^{-1}	5.5	11.8	11.0	11.7	21.7	19.3	13.5
		SE	1.0	2.4	1.3	2.2	8.2	2.6	
		gdw m^{-1}	4.73	17.54	8.41	11.22	7.46	7.37	9.46
		SE	0.94	6.81	1.67	4.67	0.45	0.30	
		inds m^{-2}	3.1	6.2	5.8	5.9	12.7	8.7	7.1
		SE	0.7	0.2	1.0	0.9	5.2	2.3	
		gdw m^{-2}	2.70	6.36	4.23	5.29	4.32	3.68	4.43
		SE	0.60	0.77	0.96	1.46	0.4	0.47	
Callinectes	Marsh	inds m^{-1}	7.2	8.7	12.2	8.8	6.5	23.6	11.2
sapidus		SE	1.0	0.2	0.4	1.1	0.3	2.6	
		gdw m^{-1}	23.0	20.03	27.64	22.88	10.21	35.66	23.24
Blue Crab		SE	2.67	2.07	8.8	7.2	6.24	14.2	
		inds m^{-2}	2.3	3.4	4.2	3.6	2.0	8.1	3.9
		SE	0.3	0.7	0.4	0.8	0.1	1.4	
		gdw m^{-2}	7.56	8.21	8.56	9.03	3.02	12.7	8.18
		SE	1.11	0.11	2.29	3.3	1.8	4.68	
	Riprap	inds m^{-1}	3.2	6.3	6.0	3.9	2.5	12.5	5.7
		SE	0.8	3.6	1.1	1.8	0.6	2.5	
		gdw m^{-1}	17.8	40.58	25.0	14.91	24.52	18.69	23.58
		SE	4.97	32.67	8.16	1.84	5.98	4.68	
		Inds m^{-2}	1.8	3.1	3.7	1.9	1.5	6.3	3.0
		SE	0.4	1.3	0.6	0.7	0.3	1.4	
		gdw m^{-2}	9.69	19.09	12.03	8.02	13.84	11.15	12.3
		SE	2.81	13.64	3.67	1.58	1.77	2.75	
Palaemonetes	Marsh	inds m^{-1}	235.7	95.8	180.1	217.6	151.8	312.4	198.9
pugio		SE	46.7	9.2	51.5	87.1	45.4	58.6	
		gdw m^{-1}	18.40	9.69	20.43	24.19	8.27	14.56	15.92
Grass		SE	2.82	1.75	4.84	8.96	2.58	2.27	
Shrimp		inds m^{-2}	76.3	38.9	61.6	77.0	47.5	98.6	66.7
		SE	21.1	5.4	12.4	25.3	15.4	13.5	
		gdw m^{-2}	8.62	4.02	6.69	9.02	2.57	4.6	5.92
		SE	2.51	0.88	0.95	3.44	0.85	0.48	
	Riprap	inds m^{-1}	240.8	131.8	216.9	248.3	244.5	370.7	242.2
		SE	42.3	27.8	53.8	43.4	70.3	36.2	
		gdw m^{-1}	18.26	14.92	25.15	21.75	12.97	13.29	17.72
		SE	2.94	1.54	6.3	3.51	2.04	2.31	
		inds m^{-2}	132.8	68.0	118.7	119.8	147.5	184.2	128.5
		SE	26.1	22.2	36.1	7.8	43.5	30.8	
		gdw m^{-2}	9.95	7.64	13.64	10.71	7.62	6.73	9.38
		SE	1.74	1.76	4.34	1.17	1.26	1.6	
Fundulus	Marsh	inds m^{-1}	7.7	12.8	13.8	15.9	10.5	15.9	12.8
heteroclitus		SE	2.7	5.9	3.3	4.9	4.9	8.7	
		gdw m^{-1}	6.21	9.45	8.35	11.26	5.82	9.31	8.4
(continued)		SE	2.58	2.16	0.83	2.37	2.04	4.05	

Species	Habitat	Data	5/4-8	5/16-18	6/1-6	6/28-7/5	8/1-2	8/27-9/2	Avg.
Mummichog	Marsh	inds m^{-2}	3.1	5.0	4.3	5.4	3.3	4.7	4.3
		SE	1.4	1.8	0.8	1.0	1.6	2.3	
		gdw m^{-2}	1.92	3.32	2.71	3.55	1.81	2.81	2.69
		SE	0.77	0.36	0.68	0.83	0.63	0.98	
	Riprap	inds m^{-1}	4.6	9.2	9.0	10.7	5.8	7.3	7.8
		SE	1.2	0.8	0.5	3.0	0.7	3.0	
		gdw m^{-1}	4.53	17.25	7.64	10.96	5.15	5.11	8.44
		SE	1.00	7.06	1.29	4.81	1.33	0.52	
		inds m^{-2}	2.6	5.	4.7	5.3	3.3	2.3	3.9
		SE	0.8	0.5	0.4	1.3	0.5	0.4	
		gdw m^{-2}	2.60	6.22	3.79	5.13	2.95	2.44	3.86
		SE	0.63	0.63	0.71	1.55	0.77	0.24	
Gobiosoma Bosc Naked Goby	Marsh	inds m^{-1}	0	0.3	0.1	0	5.8	9.1	2.5
		SE		0.1	0.1		1.9	4.0	
		gdw m^{-1}	0	0.08	0.03	0	0.42	2.27	0.47
		SE		0.05	0.03		0.09	1.34	
		inds m^{-2}	0	0.1	< 0.1	0	1.8	2.8	0.8
		SE		0.1			0.7	1.3	
		gdw m^{-2}	0	0.04	0.01	0	0.13	0.67	0.14
		SE		0.03	0.01		0.03	0.38	
	Riprap	inds m^{-1}	0.2	0.6	1.6	0.8	13.0	10.6	4.5
		SE	0.1	0.4	1.0	0.6	6.8	4.6	
		gdw m^{-1}	0.09	0.29	0.76	0.22	1.3	1.71	0.73
		SE	0.66	0.25	0.49	0.18	0.77	0.87	
		inds m^{-2}	0.1	0.3	0.9	0.5	7.7	5.6	2.5
		SE	0.1	0.2	0.6	0.4	4.2	2.9	
		gdw m^{-2}	0.04	0.02	0.44	0.143	0.79	0.93	0.39
		SE	0.03	0.01	0.28	0.12	0.48	0.56	
Menidia menidia Atlantic Silverside	Marsh	inds m^{-1}	1.0	0.6	1.4	1.4	1.3	2.3	1.3
		SE	0.8	0.3	0.4	1.0	1.3	1.6	
		gdw m^{-1}	<0.01	0.01	0.14	0.16	0.28	0.78	0.23
		SE		0.01	0.08	0.10	0.28	0.55	
		inds m^{-2}	0.4	0.1	0.5	0.4	0.4	0.7	0.4
		SE	0.3	0.1	0.2	0.3	0.4	0.4	
		gdw m^{-2}	<0.01	<0.01	0.04	0.06	0.09	0.08	0.05
		SE			0.03	0.03	0.09	0.06	
	Riprap	inds m^{-1}	0	1.9	0	0.1	1.0	1.1	0.7
		SE		1.9		0.1	1.0	1.1	
		gdw m^{-1}	0	<0.01	0	<0.01	0.19	0.37	0.10
		SE					0.19	0.37	
		inds m^{-2}	0	0.9	0	<0.1	0.5	0.7	0.4
		SE		0.9			0.5	0.7	
		gdw m^{-2}	0	<0.01	0	<0.01	0.10	0.23	0.06
		SE					0.10	0.23	
Paralichthys dentatus Summer Flounder	Marsh	inds m^{-1}	4.7	0.4	0	0	0	0.1	0.9
		SE	2.08	0.25				0.1	
		gdw m^{-1}	1.52	0.09	0	0	0	0.31	0.32
		SE	0.63	0.03				0.31	
		inds m^{-2}	1.3	0.1	0	0	0	<0.1	0.2
		SE	0.56	0.1					
		gdw m^{-2}	0.42	0.03	0	0	0	0.09	0.09
(continued)		SE	0.17	0.01				0.09	

Species	Habitat	Data	5/4-8	5/16-18	6/1-6	6/28-7/5	8/1-2	8/27-9/2	Avg.
P. dentatus	Riprap	inds m^{-1}	0.41	0	0	0	0	0	0.1
		SE	0.15						
		gdw m^{-1}	0.11	0	0	0	0	0	0.02
		SE	0.04						
		inds m^{-2}	0.23	0	0	0	0	0	<0.1
		SE	0.07						
		gdw m^{-2}	0.06	0	0	0	0	0	0.01
		SE	0.02						
Fundulus majalis Striped Killifish	Marsh	inds m^{-1}	0.2	0	1.1	1.6	1.0	3.1	1.2
		SE	0.2		0.3	0.4	0.5	2.0	
		gdw m^{-1}	0.20	0	2.33	1.83	2.24	1.51	1.35
		SE	0.13		0.49	0.79	1.27	0.52	
		inds m^{-2}	0.1	0	0.2	0.4	0.3	0.9	0.3
		SE	0.04		0.1	0.1	0.2	0.5	
		gdw m^{-2}	0.05	0	0.68	0.53	0.70	0.32	0.38
		SE	0.04		0.14	0.22	0.39	0.13	
	Riprap	inds m^{-1}	0	0	0.2	0	0	0	<0.1
		SE			0.2				
		gdw m^{-1}	0	0	<0.01	0	0	0	<0.01
		SE							
		inds m^{-2}	0	0	0.2	0	0	0	0.03
		SE			0.2				
		gdw m^{-2}	0	0	<0.01	0	0	0	<0.01
Total Sportfish	Marsh	inds m^{-1}	5.35	0.51	0.63	3.40	0.34	0.40	1.77
		SE	2.0	0.26	0.40	1.89	0.24	0.18	
		gdw m^{-1}	1.60	0.38	0.33	3.12	0.97	1.45	1.31
		SE	0.60	0.32	0.23	2.17	0.83	0.75	
		inds m^{-2}	1.38	0.15	0.13	0.95	0.11	0.39	0.52
		SE	0.56	0.08	0.09	0.51	0.08	0.05	
		gdw m^{-2}	0.44	0.04	0.04	0.76	0.33	0.40	0.34
		SE	0.42	0.01	0.01	.58	0.28	0.21	
	Riprap	inds m^{-1}	0.42	0	0	0.06	0.50	0.08	0.18
		SE	0.13			0.06	0.37	0.08	
		gdw m^{-1}	0.11	0	0	0.02	0.04	0.15	0.05
		SE	0.04			0.02	0.04	0.15	
		inds m^{-2}	0.23	0	0	0.03	0.25	0.04	0.09
		SE	0.07			0.03	0.18	0.04	
		gdw m^{-2}	0.06	0	0	0.01	0.02	0.06	0.03
		SE	0.02			0.01	0.02	0.06	

Table 14. Fringe Marsh and Riprap Site Comparison:
Mean Nekton Abundance per Meter and Mean Density per Square Meter.

Results of Kruskal-Wallis tests for differences in nekton abundance per meter (inds · m^{-1}) and density per square meter (inds · m^{-2}) within fringing marsh and riprap study sites in Sarah Creek are shown. Species with 25 or more individuals captured in the marsh or riprap were tested. Sample means from each study site were pooled for analysis (N site A = 14, B = 9, C = 8). Tukey's multiple comparison results are also shown for species that had significant Krustal-Wallis test results. Bars connecting site abbreviations (over or under) denote statistically similar ($p > 0.05$) abundance · m^{-1} or density · m^{-2} between sites.

Table 14. Fringe Marsh and Riprap Site Comparison: Mean Nekton Abundance per meter ($\cdot$ m^{-1}) and per square meter ($\cdot$ m^{-2}).

Species	A Inds · m^{-1} (SE)	B Inds · m^{-1} (SE)	C Inds · m^{-1} (SE)	P Value	A Inds · m^{-2} (SE)	B Inds · m^{-2} (SE)	C Inds · m^{-2} (SE)	P Value
Fringe Marsh					**Fringe Marsh**			
Fundulus heteroclitus	17.2 (3.1)	9.3 (1.8)	9.7 (1.8)	0.145	5.2 (0.9)	3.5 (0.9)	3.5 (0.7)	0.307
Gobiosoma bosc	0.3 (0.2)	4.1 (3.0)	3.1 (1.3)	0.123	0.1 (0.1)	1.3 (0.9)	0.9 (0.4)	0.085
Menidia menidia	1.9 (0.9)	1.1 (0.6)	0.8 (0.4)	0.764	0.6 (0.3)	0.3 (0.2)	0.4 (0.2)	0.766
Fundulus majalis	1.5 (0.8)	0.9 (0.4)	0.8 (0.3)	0.999	0.4 (0.2)	0.3 (0.1)	0.3 (0.1)	0.973
Total fish	24.8 (4.0)	16.4 (3.9)	15.5 (2.7)	0.224	7.4 (1.0)	5.7 (1.2)	5.6 (1.0)	0.301
Callinectes sapidus	10.0 (1.4)	13.2 (2.3)	10.4 (2.5)	0.377	3.0 (0.4)	5.9 (1.1)	3.8 (0.7)	*0.015
Palaemonetes pugio	265.6 (34.7)	171.1 (35.2)	115.7 (25.7)	*0.015	81.7 (10.5)	72.1 (12.4)	40.0 (7.4)	*0.025

Palaemonetes pugio C B A

Callinectes sapidus A C B

Palaemonetes pugio C B A

Species	A Inds · m^{-1} (SE)	B Inds · m^{-1} (SE)	C Inds · m^{-1} (SE)	P Value	A Inds · m^{-2} (SE)	B Inds · m^{-2} (SE)	C Inds · m^{-2} (SE)	P Value
RipRap					**Riprap**			
Fundulus heteroclitus	7.4 (1.3)	9.1 (1.6)	5.4 (0.7)	0.308	4.2 (0.7)	3.9 (0.7)	3.1 (0.4)	0.538
Gobiosoma bosc	1.0 (0.5)	2.2 (0.8)	7.8 (3.0)	*0.007	0.5 (0.5)	0.9 (0.8)	4.6 (1.8)	*0.007
Menidia menidia	0.5 (0.5)	1.4 (1.4)	0.4 (0.3)	0.504	0.3 (0.3)	0.7 (0.7)	0.2 (0.2)	0.509
Total fish	9.2 (1.3)	12.8 (2.9)	14.5 (3.4)	0.371	5.2 (0.7)	5.6 (1.0)	8.6 (2.1)	0.221
Callinectes sapidus	3.7 (0.6)	10.5 (1.2)	4.7 (1.5)	*0.002	2.1 (0.3)	4.4 (0.5)	2.8 (0.9)	*0.009
Palaemonetes pugio	259.9 (34.8)	181.9 (44.4)	232.3 (37.1)	0.331	139.9 (17.9)	76.0 (17.9)	136.9 (20.7)	0.063

Callinectes sapidus A C B

Gobiosoma bosc A B C

Callinectes sapidus A C B

Gobiosoma bosc A B C

*denotes a significant difference at $p < 0.05$

Table 15. Fringe Marsh and Riprap Site Comparison:
Mean Nekton Biomass per Meter and Mean Biomass per Square Meter.

Results of Kruskal-Wallis tests for differences in nekton biomass per meter (GDW · m^{-1}) and nekton biomass per square meter (GDW · m^{-2}) within fringing marsh and riprap study sites in Sarah Creek are shown. Species with greater than 25 individuals captured in the marsh or riprap were tested. Sample means from each sampled tide were pooled for each site (N site A = 14, B = 8, C = 9, except for *P. Pugio* where N site B = 7). Tukey's multiple comparison results are also shown for species that had significant Kruskal-Wallis test results. Bars connecting site letters (over or under) denote statistically similar ($p > 0.05$) biomass · m^{-1} or · m^{-2} between sites.

Table 15. Fringe marsh and Riprap Site Comparison: Mean Nekton Biomass per meter ($\cdot$ m^{-1}) and per square meter ($\cdot$ m^{-2}).

Species	A GDW · m^{-1}	(SE)	B GDW · m^{-1}	(SE)	C GDW · m^{-1}	(SE)	P Value	A GDW · m^{-2}	(SE)	B GDW · m^{-2}	(SE)	C GDW · m^{-2}	(SE)	P Value
Fringe Marsh								**Fringe Marsh**						
F. heteroclitus	9.82	(1.68)	7.72	(1.90)	6.76	(1.11)	0.444	2.92	(0.45)	2.83	(0.62)	2.62	(0.60)	0.756
Gobiosoma bosc	0.04	(0.03)	0.49	(0.28)	0.69	(0.52)	0.101	0.01	(0.01)	0.17	(0.09)	0.20	(0.15)	0.094
Menidia menidia	0.32	(0.26)	0.13	(0.12)	0.17	(0.10)	0.935	0.09	(0.07)	0.05	(0.05)	0.06	(0.04)	0.912
Fundulus majalis	0.70	(0.28)	1.36	(0.75)	1.29	(0.55)	0.726	0.19	(0.08)	0.38	(0.21)	0.41	(0.18)	0.689
Total Fish	12.58	(2.15)	11.15	(2.81)	9.65	(0.83)	0.691	3.68	(0.56)	4.05	(0.93)	3.54	(0.49)	0.904
Callinectes sapidus	24.37	(2.86)	25.58	(4.89)	16.76	(6.02)	0.057	7.61	(0.86)	11.04	(2.57)	6.0	(1.86)	0.093
Palaemonetes pugio	20.72	(2.58)	15.35	(3.44)	8.74	(1.42)	*0.007	6.41	(0.76)	6.48	(1.52)	3.18	(0.45)	*0.019
Palaemonetes pugio C B A								*Palaemonetes pugio* C A B						
RipRap								**Riprap**						
F. heteroclitus	6.01	(1.0)	10.44	(2.33)	5.64	(0.76)	0.205	3.46	(0.60)	4.50	(1.02)	3.30	(0.44)	0.638
Gobiosoma bosc	0.31	(0.11)	0.26	(0.20)	1.26	(0.39)	*0.021	0.17	(0.06)	0.10	(0.08)	0.77	(0.23)	*0.012
Menidia menidia	0.16	(0.16)	< 0.01		0.07	(0.06)	0.608	0.10	(0.10)	< 0.01		0.04	(0.04)	0.319
Total Fish	6.53	(0.99)	10.79	(2.24)	7.26	(0.70)	0.083	3.75	(0.58)	4.65	(0.98)	4.28	(0.37)	0.729
Callinectes sapidus	15.34	(2.83)	40.33	(12.63)	20.94	(3.04)	0.131	8.44	(1.50)	18.17	(5.59)	12.37	(1.47)	0.100
Palaemonetes pugio	18.39	(2.87)	14.90	(2.66)	18.87	(3.09)	0.633	10.00	(1.52)	6.25	(1.06)	11.18	(1.83)	0.102
Gobiosoma bosc A B C								Gobiosoma bosc B A C						

GDW = grams dry weight

*denotes a significant difference at the $\propto = 0.05$ level.

Table 16. Diel Use of Fringing Marsh and Riprap Habitat:
Mean Nekton Abundance per Meter and Mean Density per Square Meter.

Results of Mann-Whitney tests for differences in abundance of nekton per meter (Inds · m^{-1}) and abundance per square meter (Inds · m^{-2}) between day and night samples. Mean results from the eleven night tides sampled were pooled for analysis. Mean results from the eleven closest daytime tides corresponding to the site and sampling period of each night tide were also pooled for comparison. Total individuals captured during the 11 night samples and 11 daytime samples and mean number of individuals (SE) is reported. Abundance results from individual fringe marsh flume nets during the 1st sampling period were used for summer flounder analysis as 85 out of the 98 flounder captured during the entire study were captured in fringe marsh flume nets during the 1st sampling period.

Table 16. Diel Use of Intertidal Habitats: Mean Nekton Abundance per meter ($\cdot$ m^{-1}) and per square meter ($\cdot$ m^{-2}) .

Species	Day n	Day Inds $\cdot$ m^{-1} (SE)	Night n	Night Inds $\cdot$ m^{-1} (SE)	P Value	Day Inds $\cdot$ m^{-2} (SE)	Night Inds $\cdot$ m^{-2} (SE)	P Value
Fringe Marsh						**Fringe Marsh**		
Fundulus heteroclitus	351	14.1 (3.7)	353	12.8 (2.5)	0.870	5.0 (1.0)	3.9 (0.7)	0.380
Gobiosoma bosc	28	1.4 (0.7)	69	3.1 (2.3)	0.694	0.5 (0.3)	1.0 (0.7)	0.594
Menidia menidia	16	0.5 (0.4)	75	*3.0 (1.0)	0.010	0.2 (0.2)	*1.0 (0.3)	0.008
Fundulus majalis	31	1.5 (1.0)	29	1.2 (0.3)	0.341	0.4 (0.9)	0.3 (0.3)	0.350
P. dentatus T1	1	0.1 (0.1)	84	*9.3 (1.5)	0.001	0.1 (0.1)	*2.6 (0.4)	<0.001
Total Fish	470	19.1 (4.9)	654	23.9 (3.8)	0.309	6.5 (1.3)	7.2 (1.1)	0.730
Callinectes sapidus	209	9.1 (1.7)	310	12.0 (1.8)	0.088	3.4 (0.6)	4.3 (0.9)	0.470
Palaemonetes pugio	5,108	204.0 (43.0)	5,822	219 (39.0)	0.948	75.4 (13.5)	69.0 (11.3)	0.720
Riprap						**Riprap**		
Fundulus heteroclitus	209	7.0 (0.9)	152	5.6 (1.0)	0.456	4.5 (0.7)	3.0 (0.6)	0.12
Gobiosoma bosc	28	1.5 (1.5)	101	4.5 (2.6)	0.272	0.8 (0.4)	2.4 (1.5)	0.43
Total Fish	250	10.7 (1.6)	289	11.6 (3.2)	0.599	5.7 (0.8)	6.2 (1.9)	0.69
Callinectes sapidus	134	5.3 (1.2)	136	5.6 (1.5)	0.921	2.7 (0.5)	2.6 (0.6)	0.91
Palaemonetes pugio	5,685	222.5 (38.2)	6,115	230.4 (37.2)	0.793	119.7 (20.8)	117.0 (19.5)	0.93

* denotes significantly higher at the $\propto = 0.05$ level
T1 = tested results from the 1st sampling period only.

Table 17. Diel Biomass in Fringing Marsh and Riprap Habitat:
Mean Nekton Biomass per Meter and Mean Biomass per Square Meter.

Results of Mann-Whitney tests for differences in mean nekton biomass per meter (GDW $\cdot$ m^{-1}) and biomass per square meter (GDW $\cdot$ m^{-2}) in intertidal fringing marsh and riprap between day and night high tides are shown. Mean results from the eleven night tides sampled were pooled for analysis. Mean results from eleven daytime tides closest to the corresponding site and sample date of each night tide were also pooled for comparison. Species with 25 or more individuals captured in fringe marsh or riprap over the 11 day or 11 night tides used for this analysis were tested. Biomass results from individual fringe marsh flume nets during the 1st sampling period were used for summer flounder analysis as 85 out of the 98 flounder captured during the entire study were captured in fringe marsh flume nets during the 1st sampling period.

Table 17. Diel Biomass in Intertidal Habitats: Mean Nekton Biomass per meter ($\cdot$ m^{-1}) and per square meter ($\cdot$ m^{-2}).

Species	Day GDW $\cdot m^{-1}$ (SE)	Night GDW $\cdot m^{-1}$ (SE)	P Value	Day GDW $\cdot m^{-2}$ (SE)	Night GDW $\cdot m^{-2}$ (SE)	P Value
		Fringe Marsh			**Fringe Marsh**	
Fundulus heteroclitus	7.43 (1.73)	8.72 (1.74)	0.743	2.80 (0.52)	2.57 (0.44)	0.948
Gobiosoma bosc	0.24 (0.14)	0.22 (0.18)	0.493	0.08 (0.05)	0.07 (0.05)	0.493
Menidia menidia	< 0.01	*0.62 (0.33)	0.027	< 0.01	*0.20 (0.09)	0.037
Fundulus majalis	1.06 (0.57)	0.95 (0.32)	0.352	0.24 (0.16)	0.29 (0.11)	0.131
Paralichthys dentatus T1	0.15 (0.15)	*2.90 (0.46)	0.001	0.06 (0.06)	*0.78 (0.11)	0.001
Total Fish	9.06 (2.26)	12.57 (2.11)	0.293	3.18 (0.63)	3.69 (0.55)	0.55
Callinectes sapidus	23.8 (3.34)	16.25 (2.54)	0.149	*9.50 (1.8)	5.09 (0.74)	0.018
Palaemonetes pugio	15.40 (3.32)	15.48 (2.40)	0.734	5.59 (1.0)	4.83 (0.64)	0.791
		Riprap			**Riprap**	
Fundulus heteroclitus	*7.85 (0.77)	4.24 (0.86)	0.003	*4.55 (0.47)	2.27 (0.52)	0.003
Gobiosoma bosc	0.27 (0.13)	0.61 (0.27)	0.389	0.15 (0.07)	0.33 (0.16)	0.34
Total Fish	*8.47 (0.98)	5.30 (0.90)	0.036	4.73 (0.44)	2.86 (0.55)	0.055
Callinectes sapidus	23.10 (7.03)	21.40 (5.20)	0.599	13.00 (3.1)	9.68 (2.2)	0.511
Palaemonetes pugio	14.70 (1.88)	15.78 (1.87)	0.473	8.56 (1.3)	8.09 (0.88)	0.910

GDW = grams dry weight
* significantly higher at the $\propto = 0.05$ level
T1 = tested results from the 1st sampling period only.

Table 18. Habitat Utilization Comparison:
Mean Nekton Abundance per Meter and Mean Density per Square Meter.

Results of Wilcoxon paired-sample tests for differences in mean nekton abundance per meter (Inds · m^{-1}) and mean nekton density per square meter (Inds · m^{-2}) captured in fringe marsh and riprap flume nets are reported. Values listed are used in the text to describe mean abundance during the study. Statistics were performed on pooled sample means (N = 31 sample tides). Statistics were also performed on individual site results (site A = 14, site B = 8, site C = 9) when site differences were present and day and night results (day = 20, night = 11) when diel differences were present. The Mann-Whitney Test was used to compare parameters between habitats on data from individual flume net samples for *P. dentatus* (n = 17 nets, day = 8, night = 9). Mean abundance reported for *P. dentatus* are for the first sampling period only (May 4 - 8, site A).

Table 18.

Habitat Utilization Comparison: Mean Nekton Abundance per meter (Inds · m^{-1}) and per square meter (Inds · m^{-2}).

Species	Results tested	Fringe Marsh Inds · m^{-1} (SE)	Riprap Inds · m^{-1} (SE)	P Value	Fringe Marsh Inds · m^{-2} (SE)	Riprap Inds · m^{-2} (SE)	P Value
Fundulus heteroclitus	Pooled	*13.0 (1.7)	7.2 (0.8)	< 0.001	4.3 (0.5)	3.8 (0.4)	0.397
Gobiosoma bosc	Pooled	2.1 (0.9)	3.3 (1.1)	0.086	0.7 (0.3)	*1.8 (0.6)	0.039
G. bosc	Site A	0.3 (0.2)	1.0 (0.5)	0.125	0.1 (0.1)	0.5 (0.2)	0.50
G. bosc	Site B	4.1 (3.0)	2.2 (0.6)	0.500	1.3 (0.9)	0.9 (0.8)	1.00
G. bosc	Site C	3.1 (1.3)	*7.8 (3.0)	0.016	0.9 (0.4)	*4.6 (1.8)	0.008
Menidia menidia	Pooled	*1.4 (0.4)	0.7 (0.4)	0.028	0.5 (0.1)	0.4 (0.2)	0.766
M. menidia	Day	0.5 (0.3)	0.6 (0.6)	0.625	0.2 (0.2)	0.3 (0.3)	0.625
M. menidia	Night	*3.0 (1.0)	1.0 (0.6)	0.004	1.0 (0.3)	0.6 (0.4)	0.097
[a]***Paralichthys dentatus***	T1 Pooled	*5.0 (1.4)	0.4 (0.2)	0.040	*1.4 (0.4)	0.2 (0.1)	0.044
[a]*P. dentatus*	T1 Day	0.1 (0.1)	0.1 (0.1)	1.00	0.1 (0.1)	0.1 (0.1)	1.00
[a]*P. dentatus*	T1 Night	*9.3 (1.5)	0.7 (0.2)	0.001	*2.6 (0.4)	0.3 (0.1)	0.001
Fundulus majalis	Pooled	*1.1 (0.4)	< 0.1	0.001	*0.3 (0.1)	< 0.1	0.001
Morone americana	Pooled	*0.4 (0.3)	< 0.1	0.031	*0.1 (0.1)	< 0.1	0.031
Leiostomus xanthurus	Pooled	*0.3 (0.1)	< 0.1	0.041	*0.1 (< 0.1)	< 0.1	0.047
Total Fish	Pooled	*19.9 (2.3)	11.7 (1.4)	< 0.001	6.4 (0.6)	6.1 (0.8)	0.297
Callinectes sapidus	Pooled	*10.9 (1.1)	5.7 (0.8)	< 0.001	*4.0 (0.4)	2.9 (0.4)	< 0.001
C. sapidus	Site A	*10.0 (1.4)	3.7 (0.6)	< 0.001	*3.0 (0.4)	2.1 (0.3)	0.011
C. sapidus	Site B	13.2 (2.3)	10.5 (1.2)	0.188	5.9 (1.1)	4.4 (0.5)	0.086
C. sapidus	Site C	*10.4 (2.5)	4.7 (1.5)	0.008	*3.8 (0.7)	2.8 (0.9)	0.027
Palaemonetes pugio	Pooled	197.7 (22.4)	231.8 (22.3)	0.059	67.4 (6.9)	*122.5 (11.8)	< 0.001
P. pugio	Site A	256.6 (34.7)	259.9 (34.8)	0.855	81.7 (10.5)	*139.9 (17.9)	0.005
P. pugio	Site B	171.1 (35.2)	181.9 (44.4)	0.461	72.1 (12.4)	76.0 (17.9)	0.742
P. pugio	Site C	115.7 (25.7)	*232.3 (37.1)	0.004	40.0 (7.4)	*136.9 (20.8)	0.004

[a] denotes use of Mann-Whitney test, all others use Wilcoxon paired rank sign test.

* significantly higher at the $\propto = 0.05$ level.

Table 19. Habitat Utilization Comparison:
Mean Nekton Biomass per Meter and Mean Biomass per Square Meter.

Results of Wilcoxon paired-sample tests for differences in mean nekton biomass per meter (GDW m^{-1}) and mean nekton biomass per square meter (GDW m^{-2}) captured in fringe marsh and riprap flume nets are reported. Values listed are used in the text to describe mean biomass during the study. Reported values are calculated from pooled sample tide averages (N = 31 for all species except *P. Pugio*, where N = 30). Individual site (N site A = 14, B = 8, C= 9) and day and night (N day = 20, night = 11) tests were performed for species that displayed significant differences in site or diel biomass. The Mann-Whitney test was used on data from individual flume net samples taken during the 1st sampling period for *P. dentatus* (n pooled = 17, n day = 8, n night = 9). Mean biomass for *P. dentatus* are from the 1st sampling period only (May 4 - 8, site A).

Table 19. Habitat Utilization Comparison: Mean Nekton Biomass per meter (GDW · m^{-1}) and per square meter (GDW · m^{-2}).

Species	Results tested	Fringe Marsh GDW · m^{-1} (SE)	Riprap GDW · m^{-1} (SE)	P Value	Fringe Marsh GDW · m^{-2} (SE)	Riprap GDW · m^{-2} (SE)	P Value
Fundulus heteroclitus	Pooled	8.39 (0.96)	7.05 (0.84)	0.250	2.81 (0.30)	*3.68 (0.39)	0.017
F. heteroclitus	Day	8.21 (1.17)	8.59 (1.07)	0.674	2.92 (0.41)	*4.46 (0.46)	0.003
F. heteroclitus	Night	*8.72 (1.74)	4.24 (0.86)	0.019	2.57 (0.44)	2.27 (0.52)	0.880
Gobiosoma bosc	Pooled	0.34 (0.17)	*0.57 (0.15)	0.027	0.11 (0.05)	*0.32 (0.09)	< 0.001
G. bosc	Site A	0.04 (0.03)	*0.31 (0.11)	0.002	0.01 (0.009)	*0.17 (0.06)	0.002
G. bosc	Site B	0.49 (0.28)	0.26 (0.20)	0.420	0.26 (0.21)	0.10 (0.08)	0.156
G. bosc	Site C	0.69 (0.52)	1.26 (0.39)	0.109	0.20 (0.15)	*0.77 (0.23)	0.008
Menidia menidia	Pooled	*0.23 (0.12)	0.09 (0.07)	0.006	0.08 (0.04)	0.06 (0.05)	0.246
[a] *Paralichthys dentatus*	T1 Pooled	*1.61 (0.42)	0.16 (0.04)	0.030	*0.39 (0.11)	0.06 (0.02)	[a] 0.048
[a] *P. dentatus*	T1 Day	0.15 (0.15)	0.05 (0.05)	1.00	0.06 (0.06)	0.02 (0.02)	[a] 1.00
[a] *P. dentatus*	T1 Night	*2.90 (0.46)	0.26 (0.10)	0.001	*0.78 (0.11)	0.09 (0.03)	[a] 0.003
Fundulus majalis	Pooled	*1.04 (0.28)	< 0.01	< 0.001	*0.30 (0.08)	< 0.01	< 0.001
Morone americana	Pooled	0.10 (0.60)	0.03 (0.03)	0.063	0.03 (0.02)	0.01 (0.01)	0.438
Leiostomus xanthurus	Pooled	*0.36 (0.22)	< 0.01	0.016	*0.11 (0.06)	< 0.01	0.016
Total Fish	Pooled	*11.36 (1.22)	7.84 (0.80)	0.006	3.73 (0.37)	4.13 (0.37)	0.408
Total Fish	Day	10.70 (1.51)	9.24 (1.02)	0.277	3.76 (0.49)	*4.83 (0.43)	0.044
Total Fish	Night	*12.57 (2.11)	5.30 (0.90)	0.007	3.69 (0.55)	2.85 (0.55)	0.148
Callinectes sapidus	Pooled	22.53 (2.53)	23.42 (3.92)	0.878	8.03 (0.96)	12.10 (1.70)	0.081
C. sapidus	Day	25.90 (3.44)	24.51 (5.46)	0.680	9.64 (1.31)	13.2 (2.42)	0.409
C. sapidus	Night	16.25 (2.53)	21.41 (5.19)	0.272	5.09 (0.74)	9.68 (2.20)	0.083
Palaemonetes pugio	Pooled	15.87 (1.72)	17.72 (1.74)	0.269	5.46 (0.57)	*9.48 (0.97)	0.002
P. pugio	Site A	20.72 (2.58)	18.39 (2.87)	0.583	6.41 (0.76)	*10.00 (1.52)	0.035
P. pugio	Site B	15.35 (3.44)	14.90 (2.66)	0.578	6.48 (1.52)	6.25 (1.06)	0.813
P. pugio	Site C	8.74 (1.42)	*18.87 (3.09)	0.004	3.18 (0.45)	*11.18 (1.83)	0.004

* significantly higher at the $\alpha = 0.05$ level.
[a] denotes use of Mann-Whitney test, all others use Wilcoxon paired sample test.

Table 20. Summary of Habitat Comparison Results.

Significant results of Wilcoxon paired sample rank sum tests between fringe marsh and riprap nekton abundance and biomass results are shown. Marsh or riprap is written in columns where significantly greater abundance or biomass was observed in that habitat ($\propto = 0.05$). Significant difference between marsh and riprap nekton parameters was not observed where the – symbol is present. Table numbers in parenthesis indicate the table in the text that fully describes results.

Species	**# inds · m^{-1} (Table 20)**	**# inds · m^{-2} (Table 20)**	**GDW · m^{-1} (Table 21)**	**GDW · m^{-2} (Table 21)**
Fundulus heteroclitus	Marsh	-	-	Riprap
Gobiosoma bosc		Riprap	Riprap	Riprap
Menidia menidia	Marsh	-	Marsh	-
Paralichthys dentatus	Marsh	Marsh	Marsh	Marsh
Fundulus majalis	Marsh	Marsh	Marsh	Marsh
Morone americana	Marsh	Marsh	-	-
Leiostomus xanthurus	Marsh	Marsh	Marsh	Marsh
Total fish	Marsh		Marsh	
Callinectes sapidus	Marsh	Marsh		-
Palaemonetes pugio	-	Riprap	-	Riprap

Table 21. Comparison of Nekton Density (inds · m^{-2}): Sarah Creek and Goodwin Island Salt Marsh Habitat

Nekton density (# inds · m^{-2}) from this study and Goodwin Island intertidal habitats are shown. Goodwin values are from a drop ring study performed by Cicchetti (1998a). Goodwin marsh edge drop rings were deployed half on and half off the marsh. Fringe marsh values were deployed 1 – 3 meters from the marsh edge. Ayers (1995) values are from a weir study performed in marshes at Goodwin Island. Marshes with an open exposure (O) and protected (P) are separated. Ayers values are inflated because means were calculated only from time periods when reported species were caught. Total fish for Ayers were created using data listed in the thesis.

Species	**Sarah Creek Marsh 2000 # inds·m^{-2}**	**Sarah Creek Riprap 2000 # inds·m^{-2}**	**Goodwin Marsh Edge 1995 # inds·m^{-2}**	**Goodwin Fringe (1-3m) 1995 # inds·m^{-2}**	**Goodwin Ayers 1994 # inds·m^{-2}**
Mummichog	4.34	3.80	0.56	1.86	9.8 (P) 1.3 (O)
Striped Killifish	0.32	0.03	0.16	0.11	0.3 (P) 0.04 (O)
Naked Goby	0.65	1.81	0.07	0.0	0.09 (P) 0.04 (O)
Atl. Silverside	0.46	0.37	0.43	0.20	0.04 (P) 2.1 (O)
Total Fish	6.42	6.07	2.32	2.43	9.23 (P) 4.26 (O)
Blue Crab	3.96	2.88	3.62	1.14	No data
(*P. Pugio*)	67.40	122.5	12.29	8.49	No data
# Fish Species	16	10	13		11

Table 22. Comparison of Biomass (gdw · m^{-2}) for Selected Nekton Species in Sarah Creek and Goodwin Island Marshes.

Nekton biomass (gdw · m^{-2}) from this study and Goodwin Island intertidal habitats (Cicchetti 1998a) is shown. Goodwin edge values are from drop rings deployed half on and half off the marsh. Goodwin fringe are values from drop rings deployed between 1-3 meters from the marsh edge. Sarah Creek values were calculated by dividing the total flume net biomass by the flood distance in each net.

Species	**Sarah Creek gdw · m^{-2}**	**Goodwin Edge gdw · m^{-2}**	**Goodwin Fringe gdw · m^{-2}**
Mummichog	2.81	0.66	0.50
Naked Goby	0.11	0.006	0
At. Silverside	0.08	0.10	0.06
Total Fish	3.73	1.23	0.82
Blue Crab	8.03	3.86	1.51
Grass Shrimp	5.46	0.50	0.34

Table 23. Comparison of Nekton Abundance and Biomass per Meter of Marsh Edge : Sarah Creek, VA vs. Newport River, N.C.

Comparison of nekton abundance and biomass captured per linear meter of marsh edge in Sarah Creek, VA and marshes in the Newport River, North Carolina (Hettler 1989). Marshes sampled in Sarah Creek flooded 3 meters from the edge on average. North Carolina marshes flooded between 15.4 to 33 meters landward from the creek edge.

Species	**Sarah Creek # inds · m^{-1}**	**Hettler 1989 # inds · m^{-1}**	**Sarah Creek GDW · m^{-1}**	**Hettler 1989 GDW · m^{-1}**
Mummichog	12.99	17.60	8.39	12.6
Naked Goby	2.08	0.19	0.34	0.11
Atl. Silverside	1.38	1.17	0.23	1.17
Total Fish	19.86	50.93	11.36	44.94
Blue Crab	10.93	0.55	22.53	18.76

Table 24. Species Length Per Sampling Period and Combined.

Median total length, mean total length and standard deviation of the mean size of *Fundulus heteroclitus* (24a), *Callinectes sapidus* (24b), *Palaemonetes pugio* (24c), *Gobiosoma bosc* (24d) and *Menidia menida* (24e) captured during each sampling period and over the entire study is listed. Mann-Whitney tests were performed on pooled data from each sampling period using the number of individuals (n) captured during that period. The Wilcoxon paired rank test statistic is given for total *Fundulus heteroclitus*, *Callinectes sapidus* and *Palaemonetes pugio* (depicted by a superscript [b]). Median length from each sample tide (N = 31) was used for the Wilcoxon paired rank test. The Mann-Whitney test was performed on total *Gobiosoma bosc* and *Menidia menidia* individuals captured because zero of these animals were captured in many of the sample tides. Total median and mean length and standard error of the mean size was calculated using the total number of individuals captured. Total size parameters therefore do not reflect the information in the Wilcoxon paired rank sign test and are descriptive only.

Table 24a. ***Fundulus heteroclitus*** **Total Length.**

Sample Period	Habitat	Median	Mean	Std. Dev.	n	P Value
1	Marsh	55	55.6	14.2	131	0.044*
	Riprap	57	58.5	14.8	73	
2	Marsh	54	46.6	24.0	151	0.000*
	Riprap	63	59.5	20.8	109	
3	Marsh	50	38.4	25.7	185	0.018*
	Riprap	56	45.2	27.6	100	
4	Marsh	34	41.0	24.2	243	0.122
	Riprap	58	45.7	27.6	136	
5	Marsh	28	37.5	24.8	52	0.004*
	Riprap	50	50.6	23.3	33	
6	Marsh	47	48.0	14.0	176	0.012*
	Riprap	62	54.1	23.3	52	
Total[b]	Marsh	50	44.4	22.43	938	[b]0.022*
	Riprap	59	51.7	24.63	503	

* denotes statistical significance at $\propto = 0.05$
[b] denotes use of Wilcoxon paired rank sum test (N = 31), all others use Mann-Whitney test.

Table 24b. ***Callinectes sapidus*** **Total Length.**

Sample Period	Habitat	Median	Mean	Std. Dev.	n	P Value
1	Marsh	41	42.2	18.3	123	0.004*
	Riprap	46	51.0	20.9	58	
2	Marsh	33	36.0	18.4	104	0.000*
	Riprap	57	53.5	25.0	75	
3	Marsh	24	32.1	22.3	145	0.009*
	Riprap	33.5	40.0	23.1	69	
4	Marsh	25	32.2	19.5	127	0.001*
	Riprap	43	46.8	25.9	46	
5	Marsh	25	25.8	17.0	34	0.003*
	Riprap	53	57.4	37.7	16	
6	Marsh	17	22.7	18.4	229	0.704
	Riprap	17	23.7	19.0	119	
Total[b]	Marsh	24	31.2	20.4	762	[b]0.0001*
	Riprap	33	40.7	25.9	383	

* denotes statistical significance at $\propto = 0.05$
[b] denotes use of Wilcoxon paired rank sum test (N = 31), all others use Mann-Whitney test.

Table 24c. ***Palaemontes pugio*** **Total Length.**

Sample Period	Habitat	Median	Mean	Std. Dev.	n	P Value
1	Marsh	30	30.3	4.2	425	0.0008*
	Riprap	28	29.4	5.2	425	
2	Marsh	32	32.2	4.7	292	0.000*
	Riprap	35	33.9	4.3	300	
3	Marsh	35	33.8	4.4	222	0.594
	Riprap	35	33.9	4.4	200	
4	Marsh	33	32.2	7.4	344	0.009*
	Riprap	32	30.2	9.0	374	
5	Marsh	24	23.8	9.5	125	0.889
	Riprap	21	23.8	9.7	149	
6	Marsh	22	23.6	6.7	249	0.037*
	Riprap	21	22.0	6.2	250	
Total[b]	Marsh	31	30.0	7.0	1657	[b]0.773
	Riprap	30	29.3	7.8	1698	

* denotes statistical significance at $\propto = 0.05$

[b] denotes use of Wilcoxon paired rank sum test, all others use Mann-Whitney test.

Table 24d. ***Gobiosoma bosc*** **Total Length.**

Sample Period	Habitat	Median	Mean	Std. Dev.	N	P Value
1	Marsh				0	Not tested
	Riprap	36.5	40.3	13.8	4	
2	Marsh	42	40.0	3.5	3	Not tested
	Riprap	43	46.4	11.2	8	
3	Marsh	43	43.0	na	1	Not tested
	Riprap	46.5	47.0	8.8	12	
4	Marsh				0	Not tested
	Riprap	41	39.5	8.6	16	
5	Marsh	20	22.7	8.9	33	0.013*
	Riprap	26	27.0	8.7	75	
6	Marsh	26	27.9	8.6	83	0.005*
	Riprap	29.5	31.4	8.9	86	
Total	Marsh	25	26.9	9.2	120	
	Riprap	30	32.0	10.7	201	

* denotes statistical significance at $\propto = 0.05$

Table 24e. ***Menidia menidia*** **Total Length.**

Sample Period	Habitat	Median	Mean	Std. Dev.	N	P Value
1	Marsh	10.5	10.8	2.1	17	Not tested
	Riprap				0	
2	Marsh	19	17.4	5.3	7	0.027*
	Riprap	11	11.5	1.9	23	
3	Marsh	26	29.9	10.4	15	Not tested
	Riprap	37	37.0	Na	1	
4	Marsh	38	37.7	7.3	26	Not tested
	Riprap				0	
5	Marsh	45	46.8	5.3	8	0.826
	Riprap	44	45.8	5.9	5	
6	Marsh	52	54.5	7.8	27	0.161
	Riprap	55	55.5	3.6	13	
Total	Marsh	38	36.0	17.1		
	Riprap	14.5	29.8	20.9		

* denotes statistical significance at $\propto = 0.05$

		Marsh			Riprap		
Species	**Measurement**	**Median**	**Mean**	**Std D**	**Median**	**Mean**	**Std D**
Paralichthys dentatus	TL	53	55.3	13.1	52.5	52.5	6.2
Fundulus majalis	TL	58	59.0	22.1	n=2	16, 16 mm	
Morone americana	TL	44.5	57.6	11.6	n=2	50, 245 mm	
Leiostomus xanthurus	TL	57	62.7	16.7	n=1	12 mm	
Symphurus plagiusa	TL	65	60.8	9.5	n=0		
Mugil curema	TL	71	71.3	32.6	n=0		
Gobiesox stumosus	TL	n=0			38.5	38.0	6.2
Morone saxatilis	TL	n=2	54,45 mm		n=0		
Luciana parva	TL	n=2	37,36 mm		n=0		
Bairdiella chrysura	TL	n=0	68,117 mm		n=0		
Cynoscion sp.	TL	n=1	>80 mm (tail eaten)		n=2	38,38 mm	
Pomatomus saltatrix	TL	n=1	40 mm		n=0		
Chaetodipterus faber	TL	n=1	42 mm		n=0		
Anguilla rostrata	TL	n=0			n=1	445 mm	

Table 25. Size of other nekton from marsh and riprap samples.

Median and mean total lengths for nekton from marsh and riprap samples are given. Two *Morone americana* individuals captured in the fringe marsh (245 and 260 mm TL) were not used in calculations for this species. All individuals captured from each habitat over the entire sampling period were pooled. One standard deviation from the mean is also given.

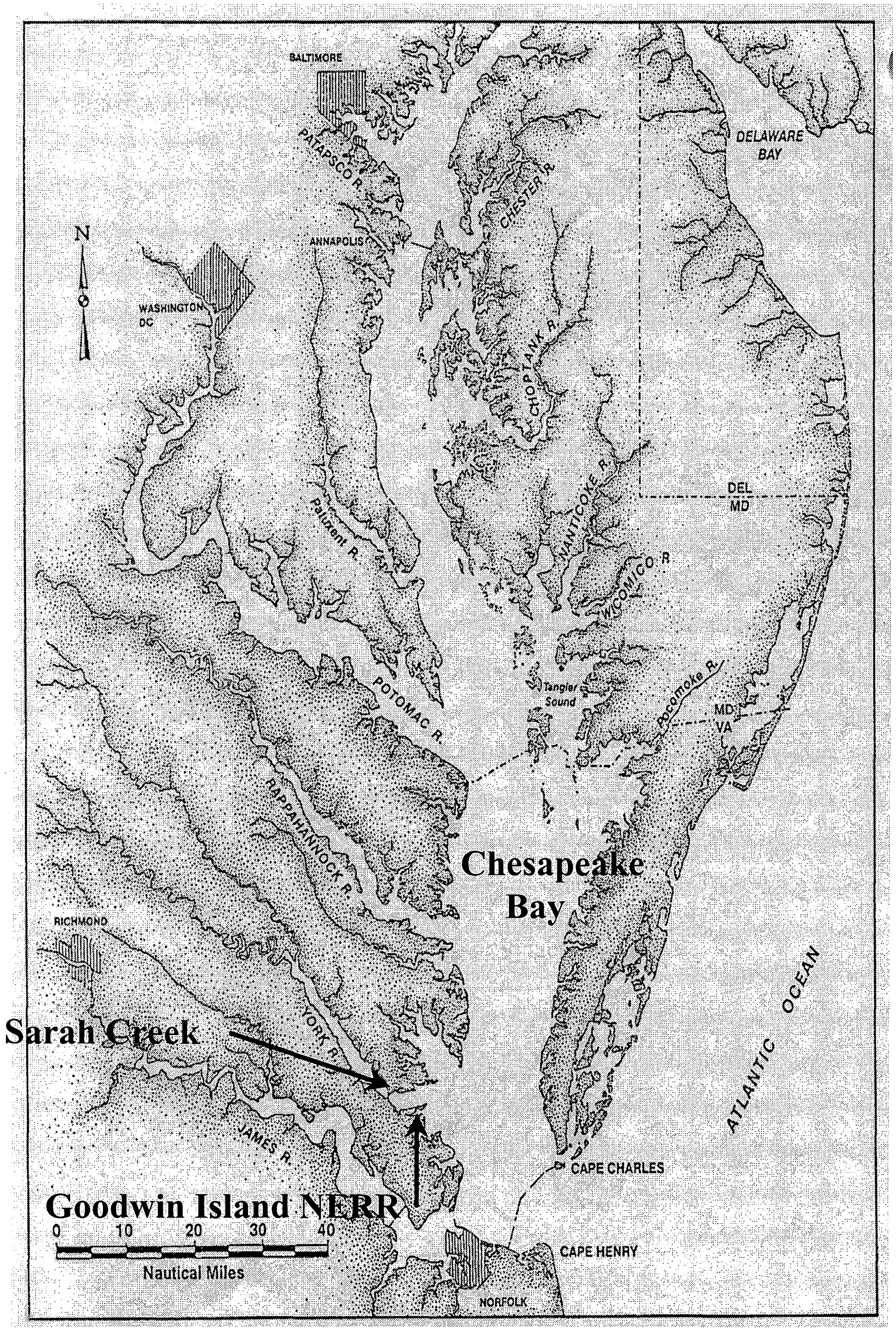


Figure 1. Map of Chesapeake Bay with Study Site Indicated

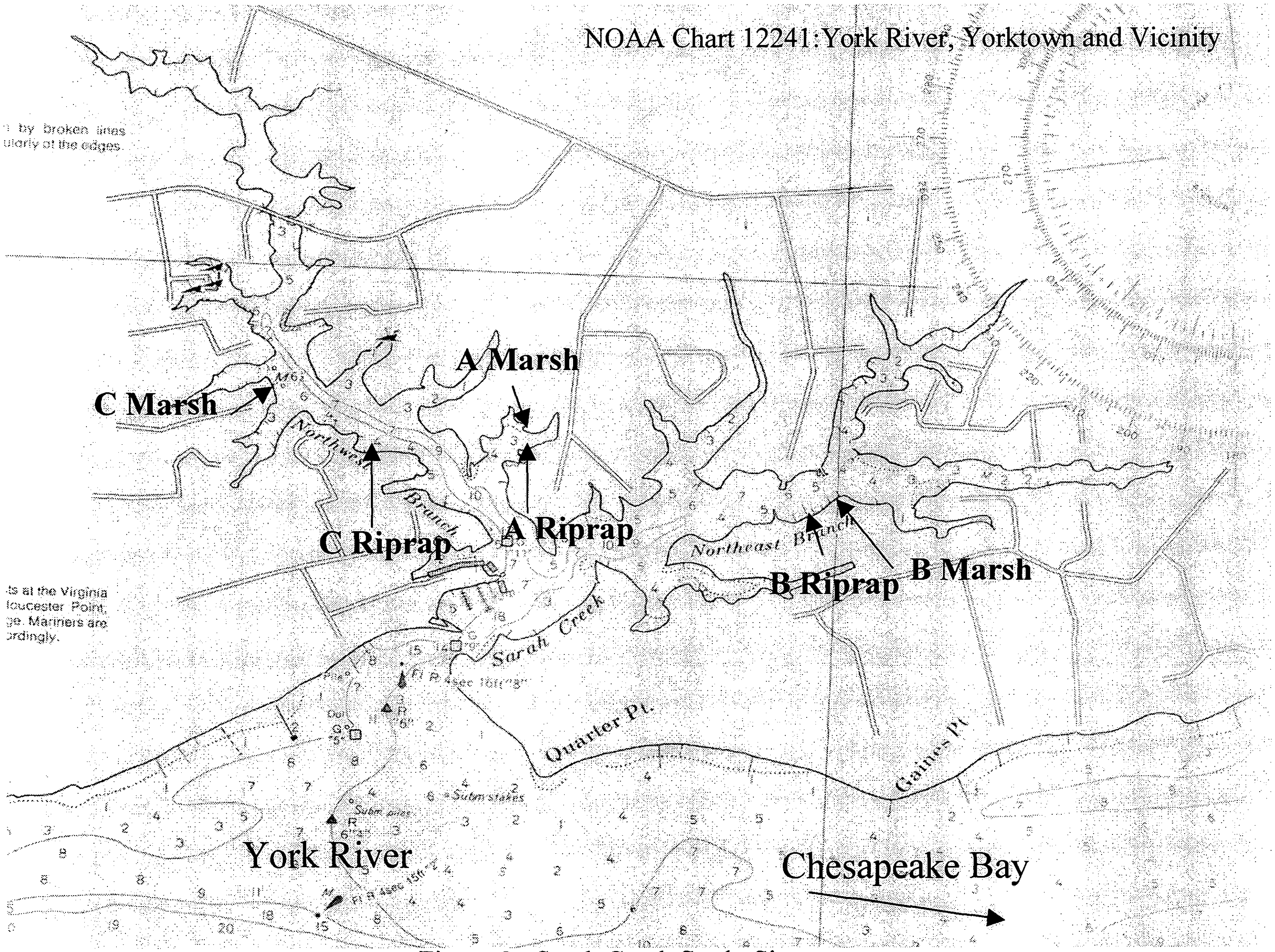


Figure 2. Sarah Creek Study Sites.
Paired study sites A, B and C are shown. Marsh indicates fringe marsh and Riprap indicates riprap sample sites

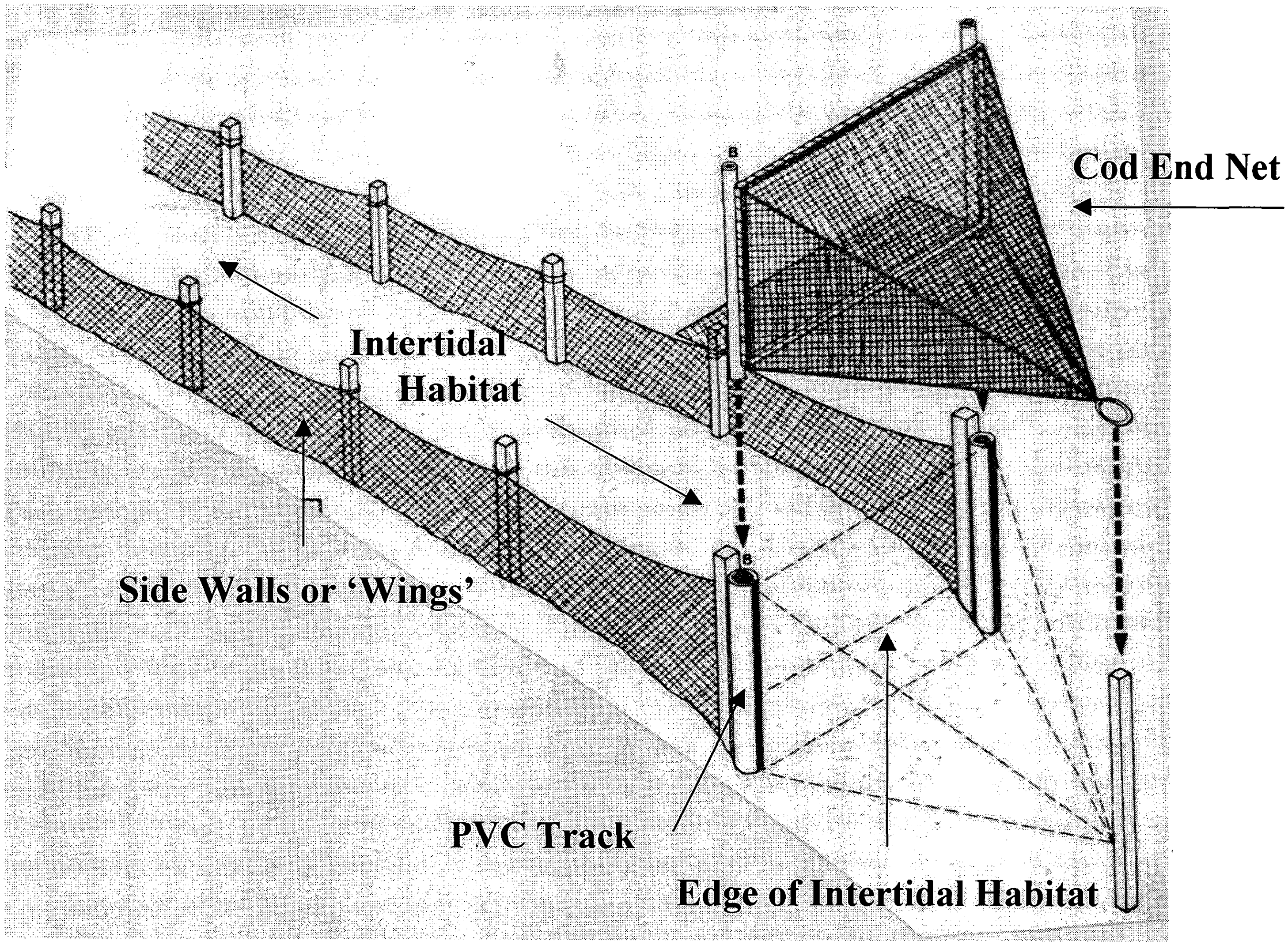


Figure 3. Flume Net Diagram (from McIvor and Odum 1986).

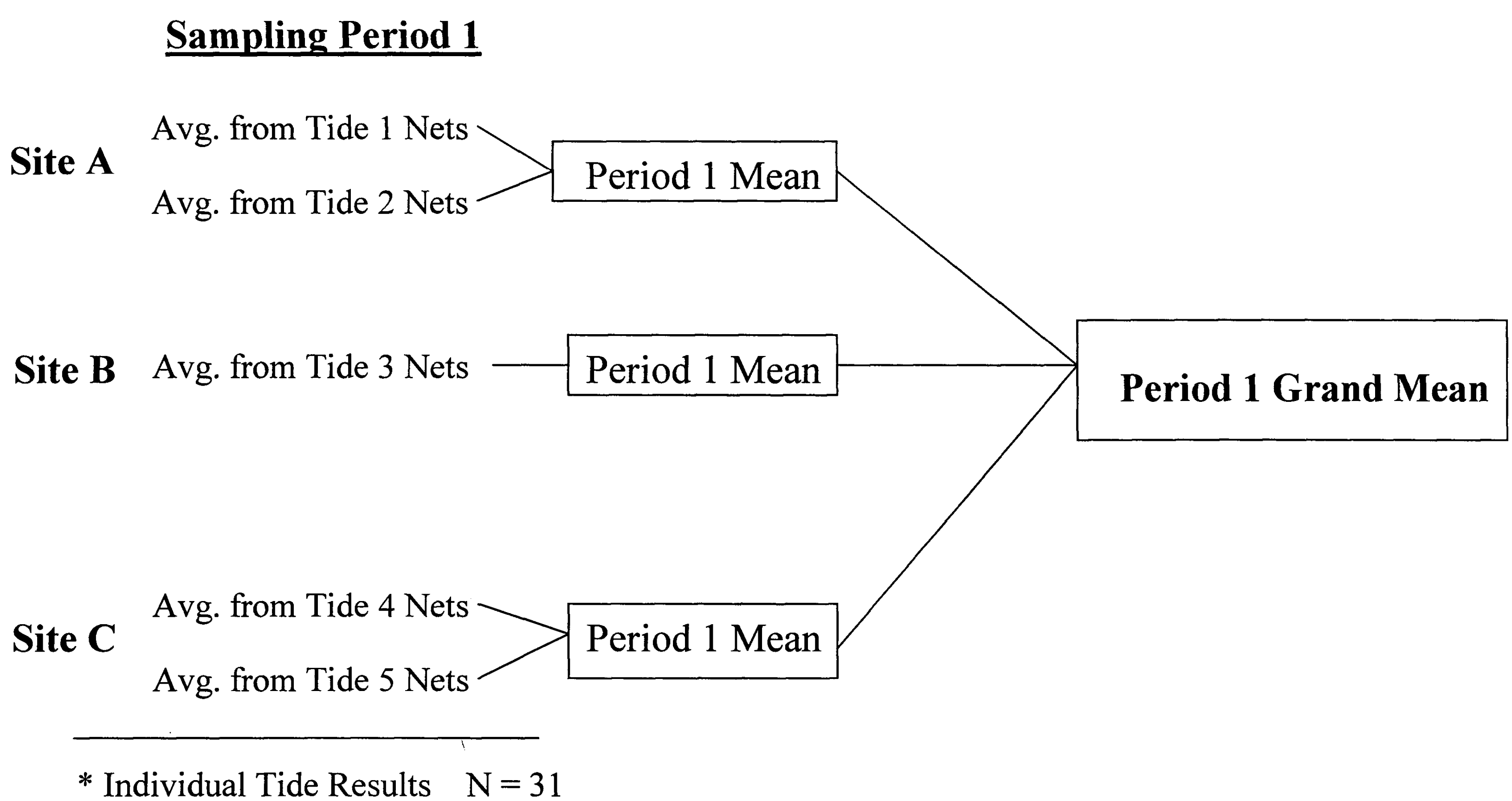


Figure 4. Calculation of means.
Average results from each tide sampled (N=31) was used in the text to describe overall abundance and biomass and for sample site, diel and paired statistical comparisons. Period means are used in the text to describe temporal trends.

Figure 5 and 6. Average Tide Height and Intertidal Area Flooded.

Average tidal height (cm) at the mouth of sampled flume nets and average intertidal area flooded (m^2) inside sampled flume nets when block nets were tripped during each sampling period. Error bars denote one standard error.

Figure 5. Average Tide Height at Net Deployment

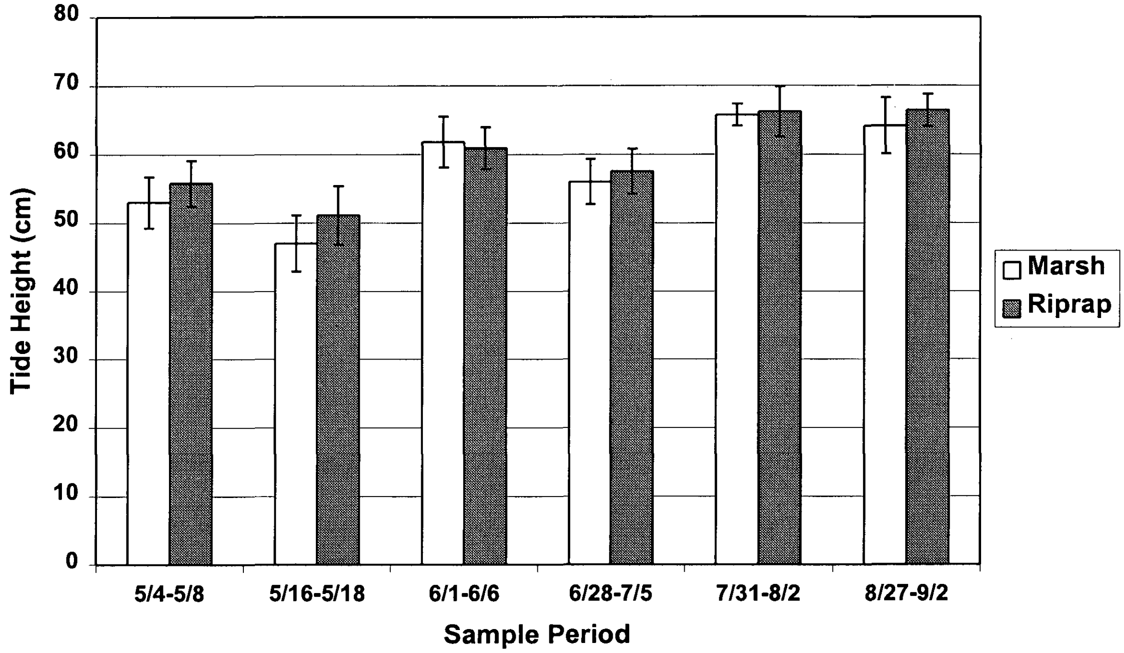


Figure 6. Average Intertidal Area Flooded

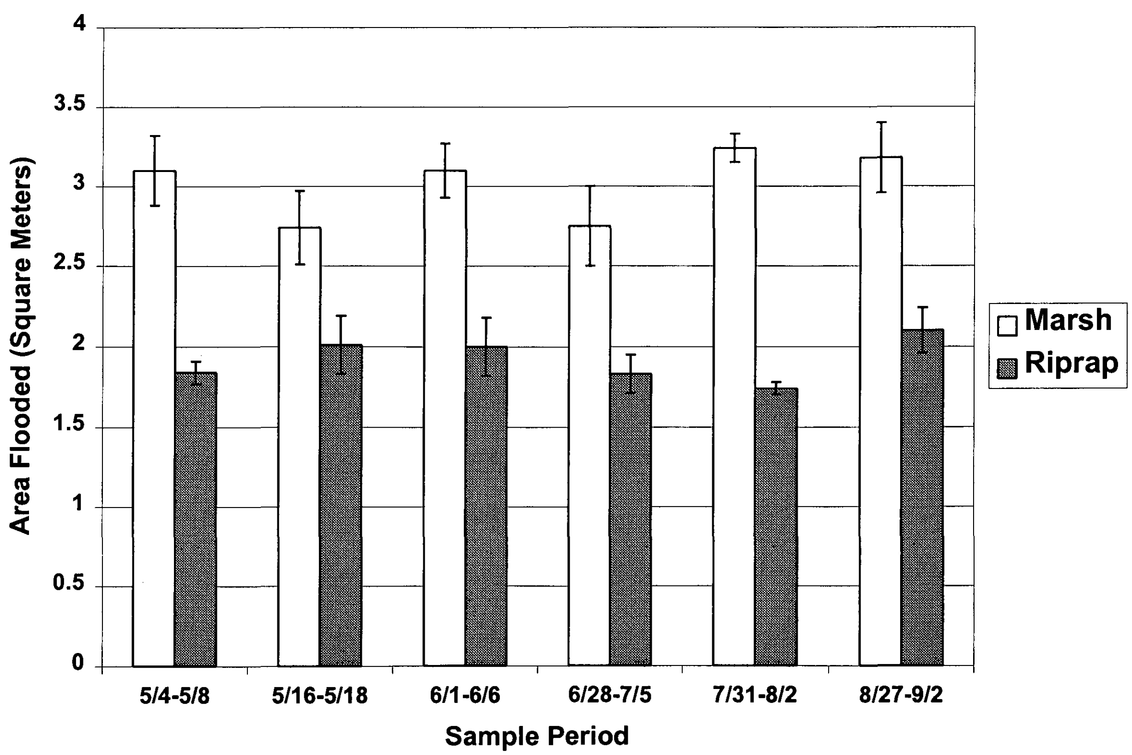


Figure 7. Mummichog Recovery Efficiency, Site A.

Recovery efficiency of mummichogs from site A fringe marsh and riprap flume nets during the first 8 weeks of flume net use. Error bars indicate one standard error. Fringe marsh estimates are from 2 trials during May 4-8 and 3 trials during May 16-18 and June 1-6. Riprap estimates are from 3 trials during May 16-18 and 2 trials during June 1- 6. Only 1 recovery trial was performed when no error bars are present.

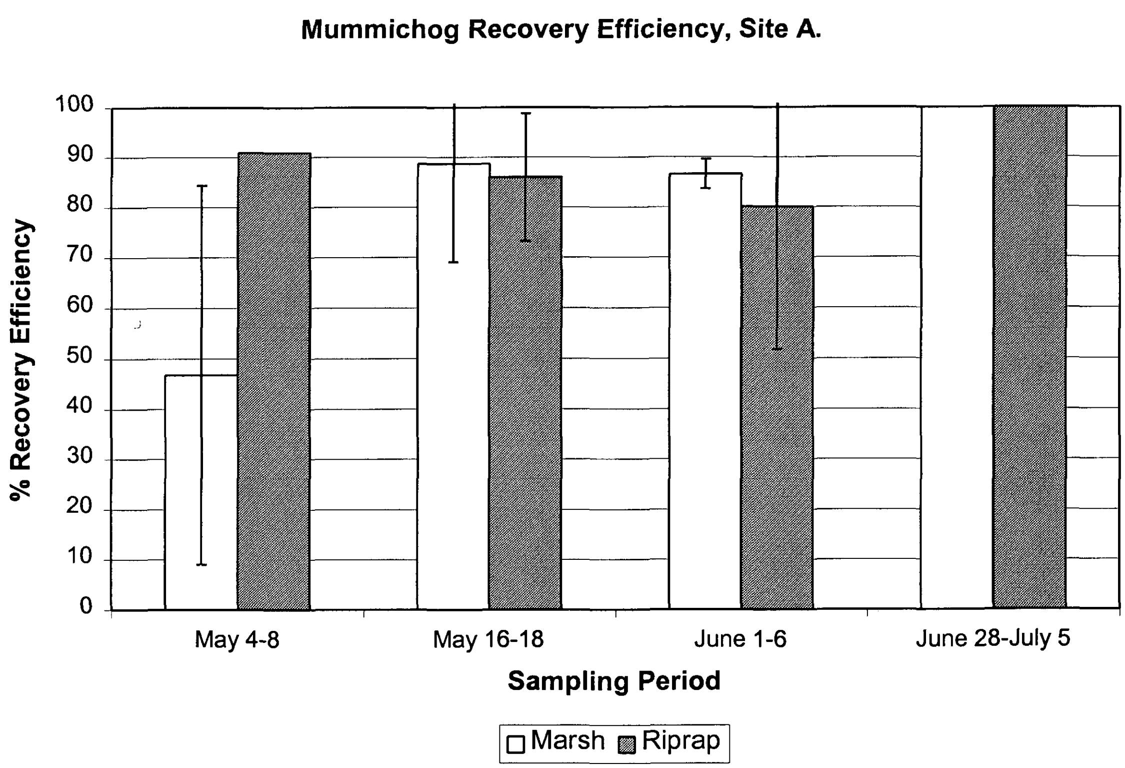


Figure 8 and 9. Relative Abundance and Biomass.

The percent of the total abundance and the total biomass of selected species captured in fringe marsh and riprap is shown. The 'Total Comm. Fish' category includes any fish that has commercial or sport value. This category combines summer flounder, white perch, spot, white mullet, striped bass, silver perch, sea trout, bluefish, spadefish and the American eel. The 'Other Fish' category includes blackcheek tonguefish, rainwater killifish, striped blenny, skilletfish, unidentified larvae, and the unidentified clupeid.

Figure 8. Relative Nekton Abundance

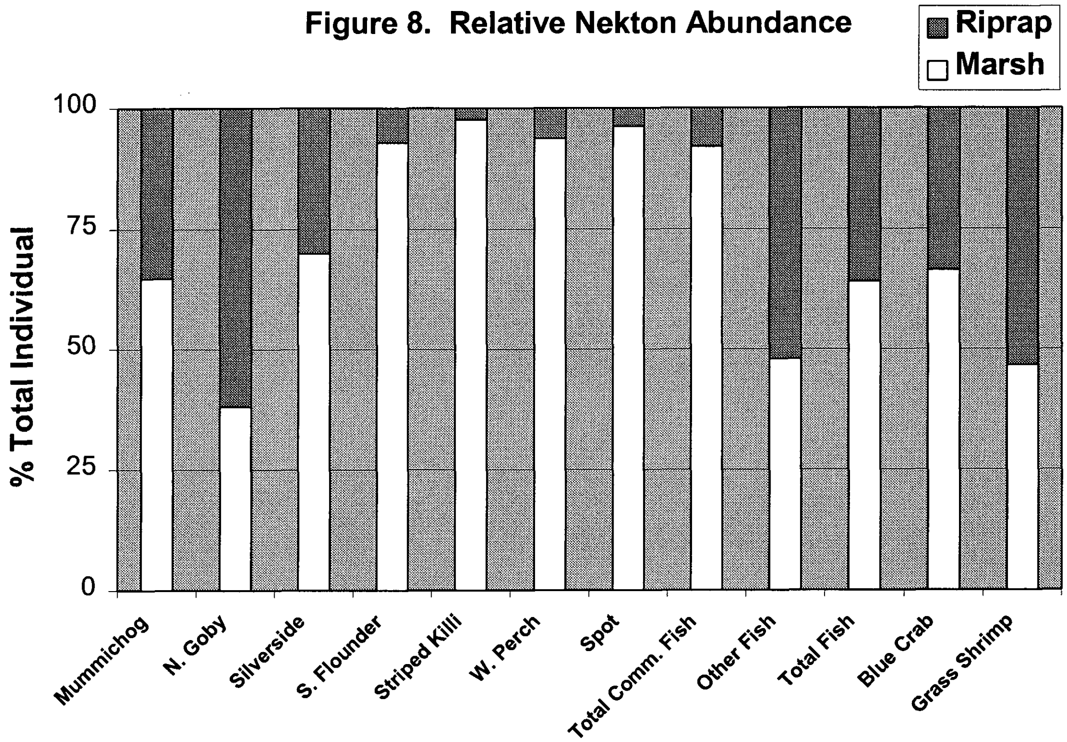


Figure 9. Relative Nekton Biomass

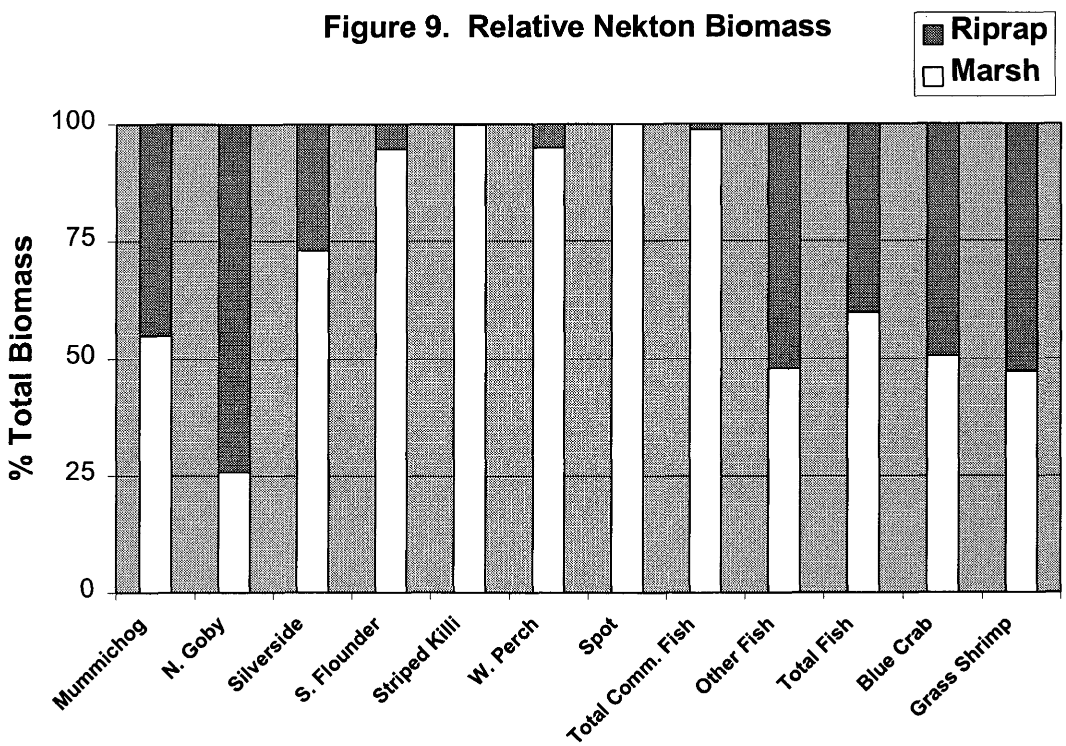


Figure 10 and 11. Cumulative Fish Abundance and Biomass Dominance Curves.

Cumulative abundance and biomass, expressed as percentage of the total fish abundance and fish biomass captured in riprap and fringe marsh, are plotted against species ranked in decreasing order of abundance and biomass. Ranks are derived from relative abundance and biomass values (Tables 11 and 12). Species ranks are not the same for fringe marsh and riprap curves, nor are they the same for the abundance and biomass figures.

Figure 10. Fish Species Abundance Dominance Curves

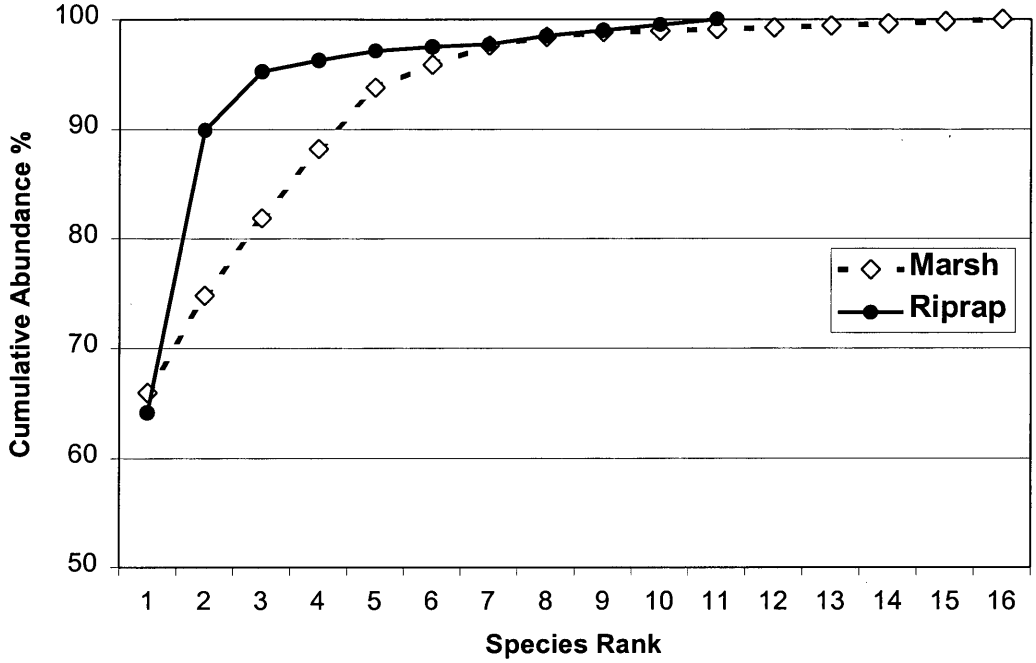


Figure 11. Fish Species Biomass Dominance Curves

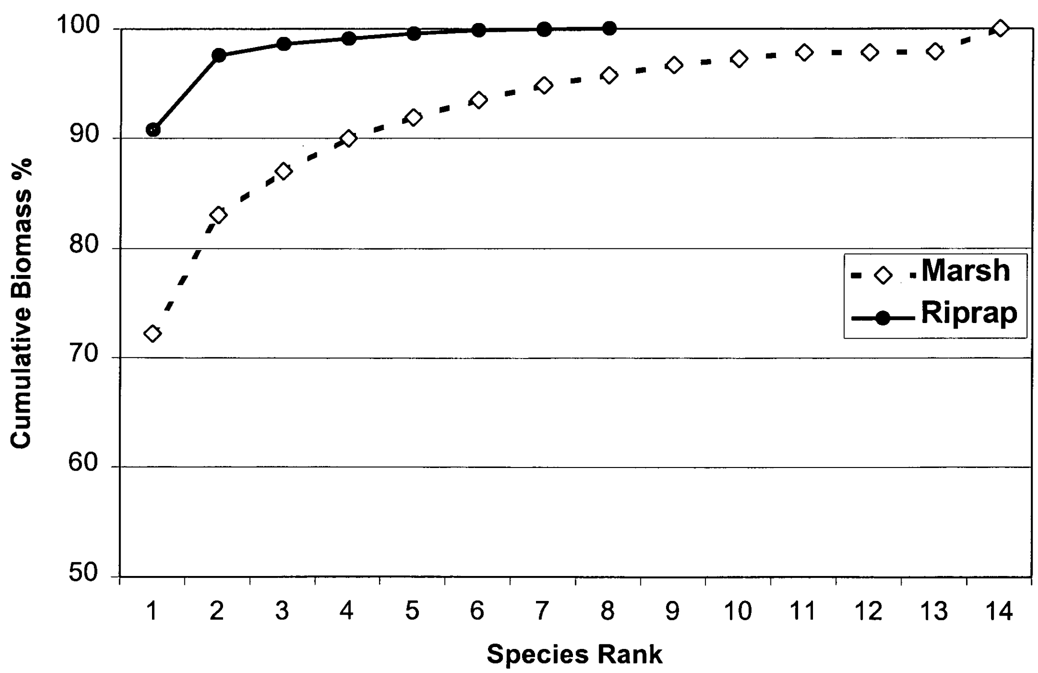


Figure 12. Relative Length Frequency for Selected Species

Relative length frequency of blue crabs (12a), mummichogs (12b) and naked gobies (12c) captured in fringe marsh and riprap. Percent frequency is the number of individuals in each size class / total individuals of that species captured in each habitat.

Figure 12a. Blue Crab Relative Length Frequency

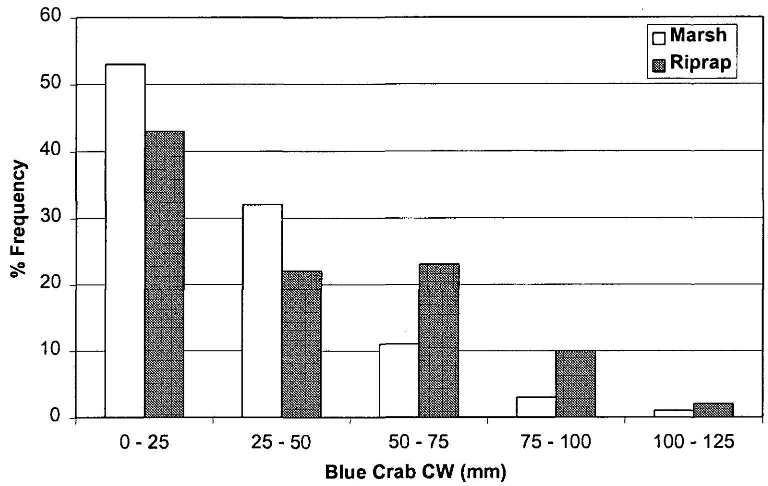


Figure 12b. Mummichog Relative Size Frequency

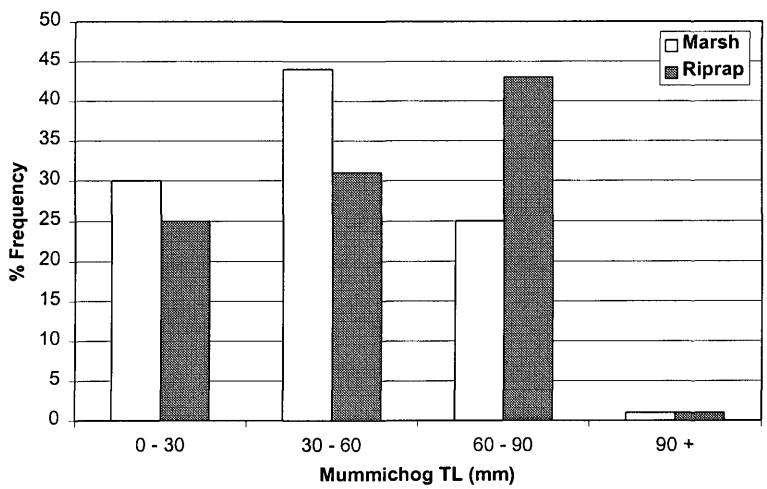


Figure 12c. Naked Goby Relative Length Frequency

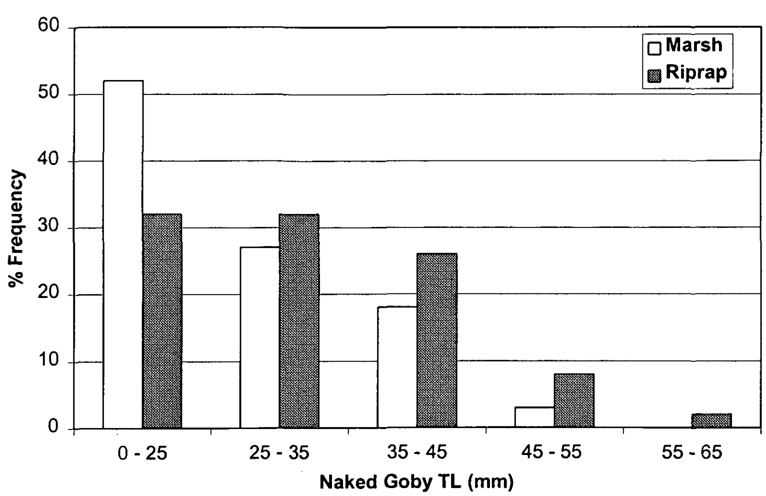
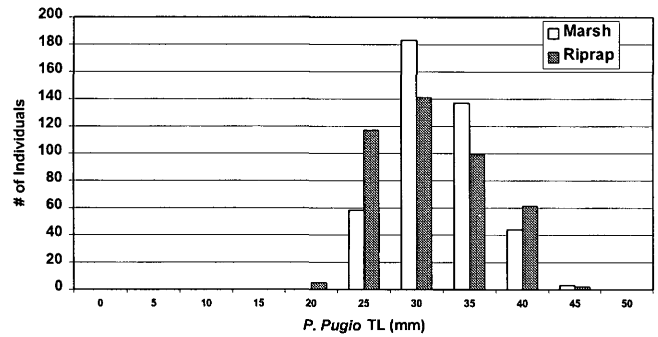


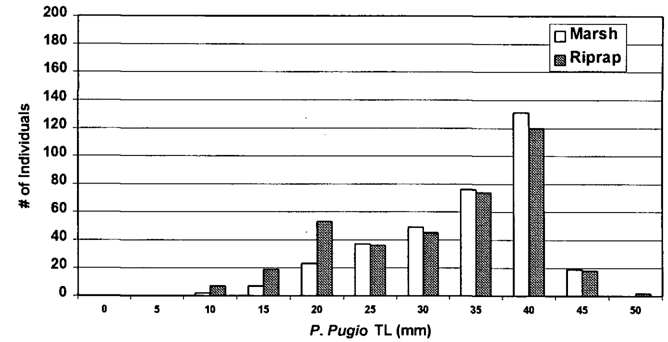
Figure 13. Length-Frequency Histograms for Selected Species

Length-frequency histograms of the five most abundant nekton species. The total number of individuals in each size bin captured during each sampling period is shown. Sampling effort was not even for each period. Grass shrimp (13a), blue crabs (13b), mummichogs (13c), naked goby (13d) and Atlantic silversides (13e) are shown in that order.

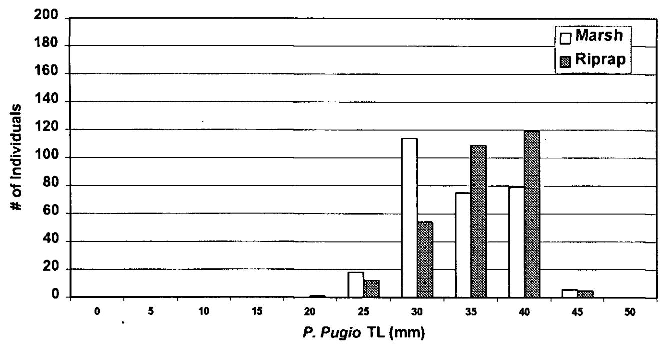
May 4 - 8



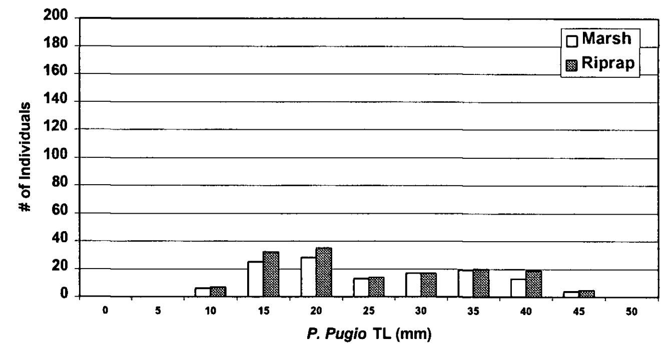
June 28 - July 5



May 16 - 18



Aug 1 - Aug 2



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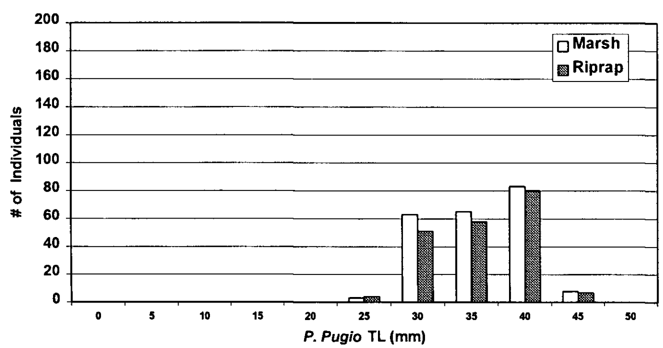

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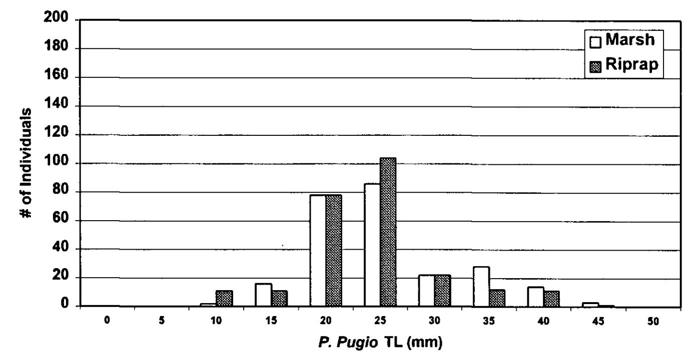

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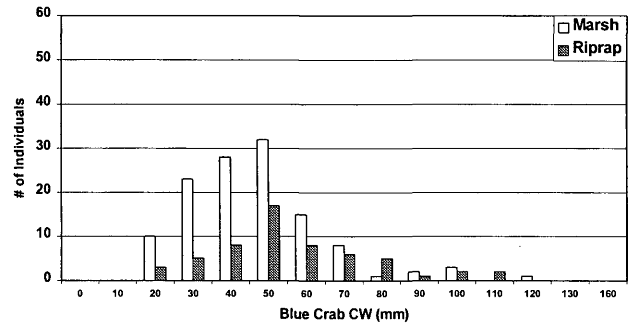

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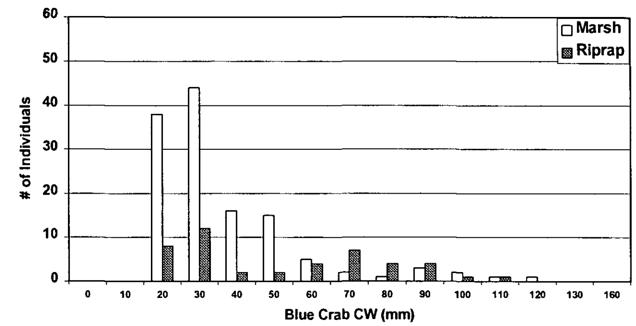

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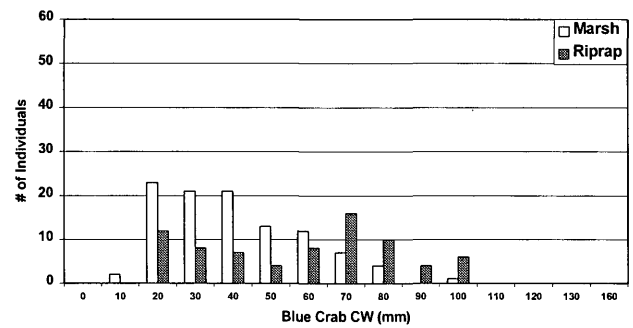

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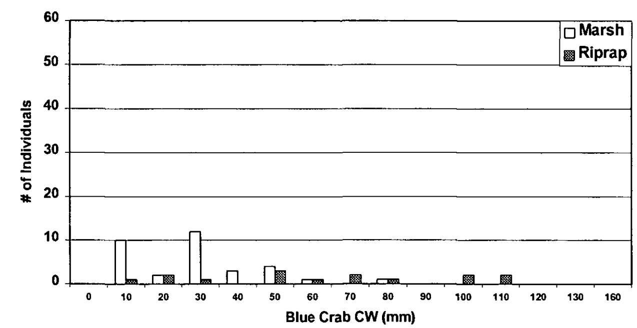

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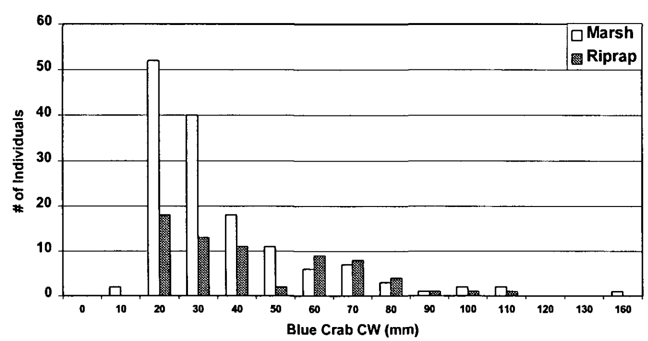

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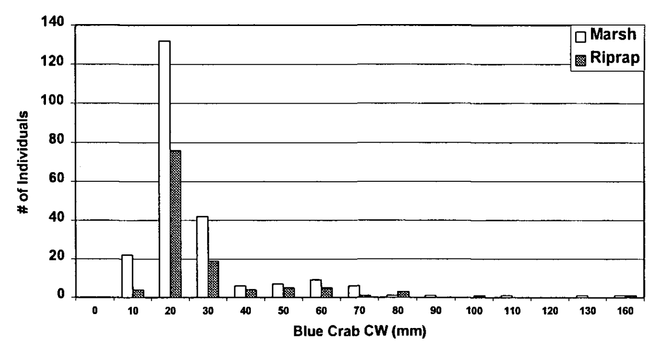

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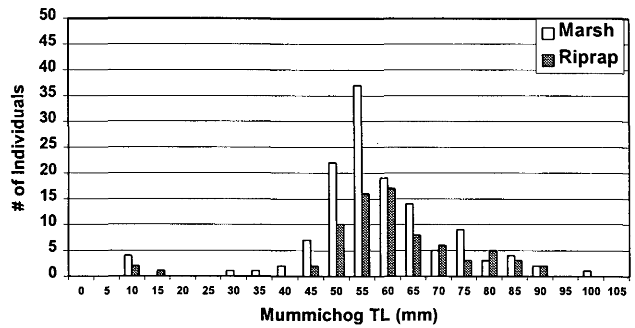

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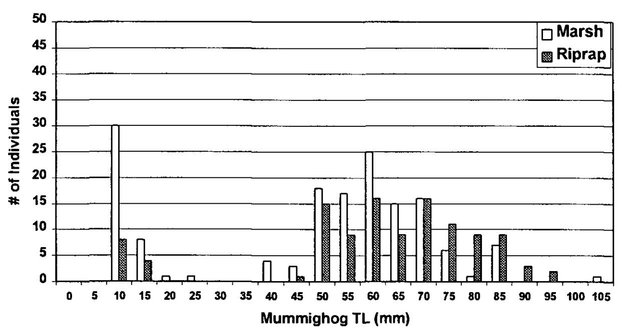

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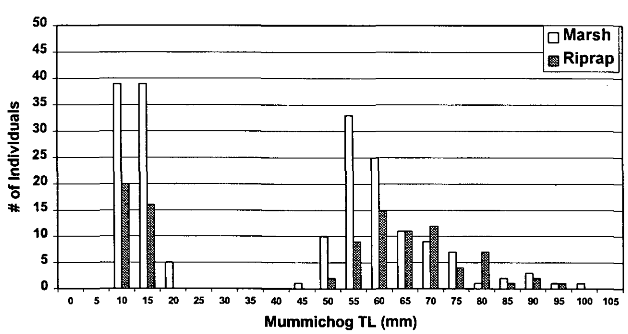

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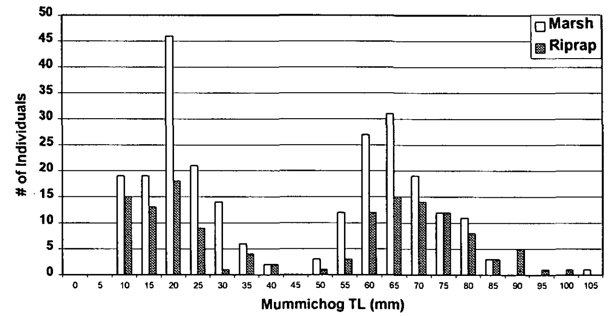

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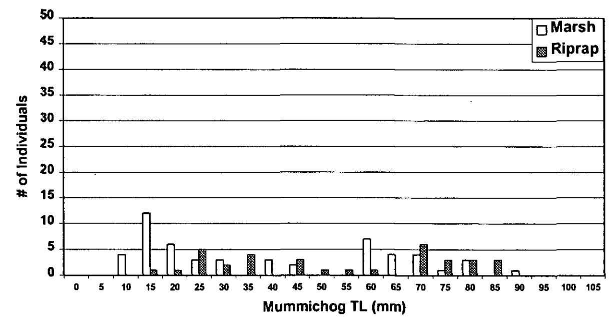

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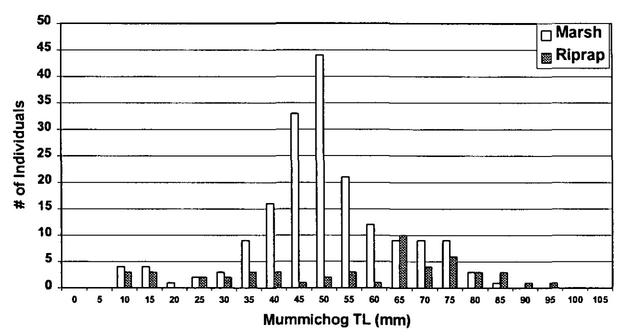

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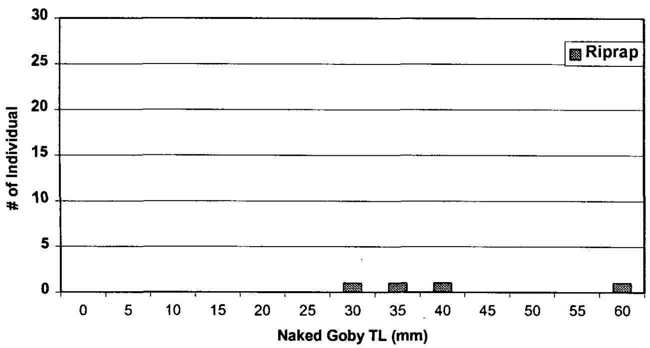

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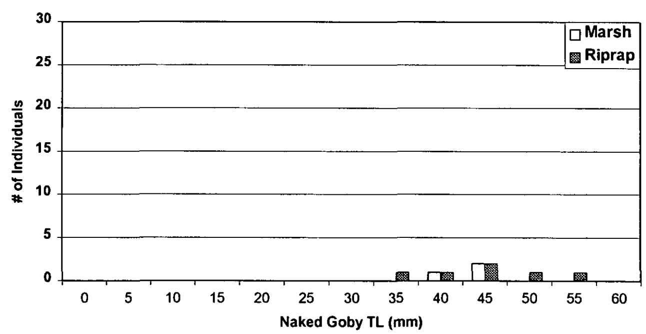

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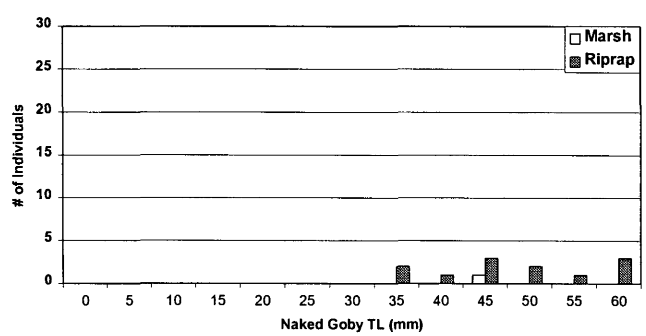

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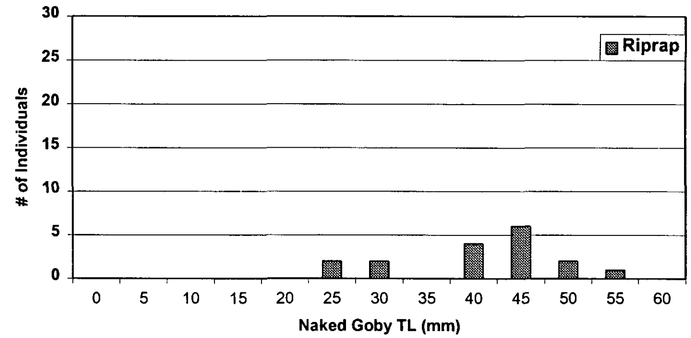

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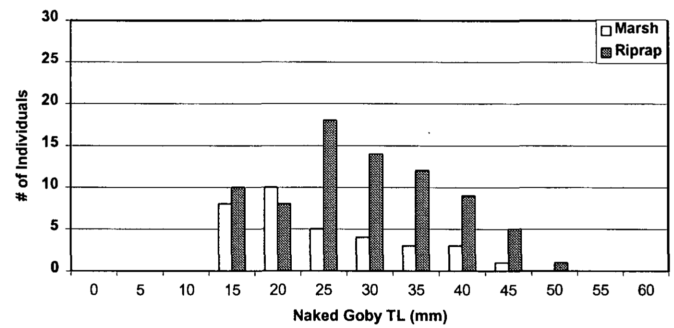

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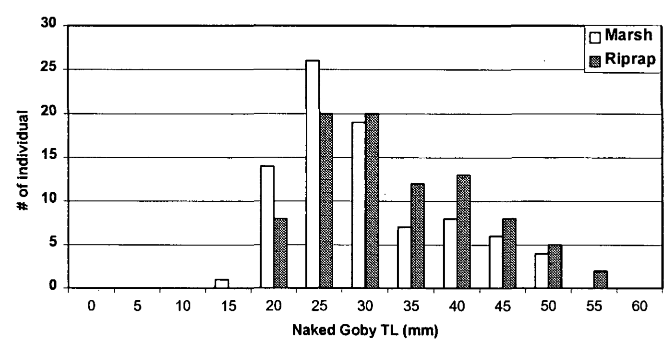

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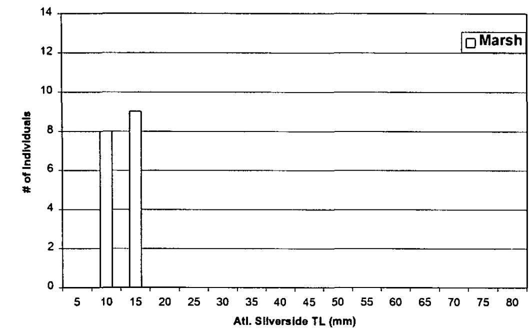

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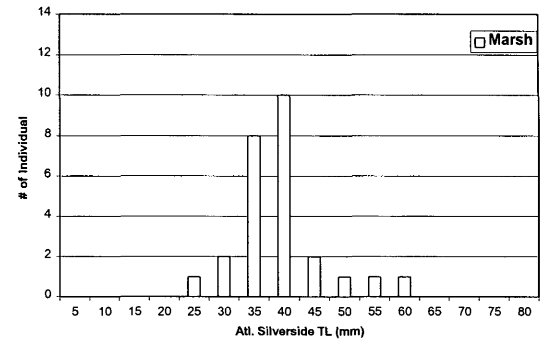

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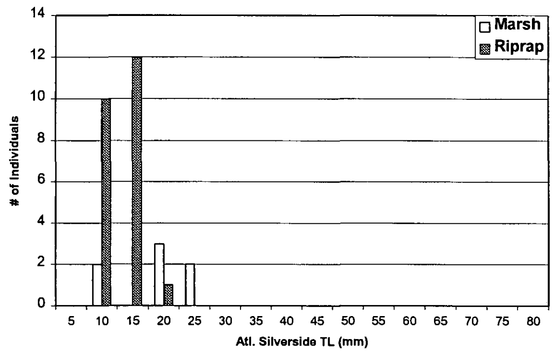
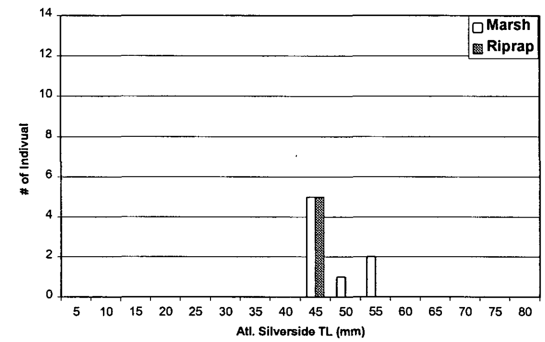


Aug 1- 2



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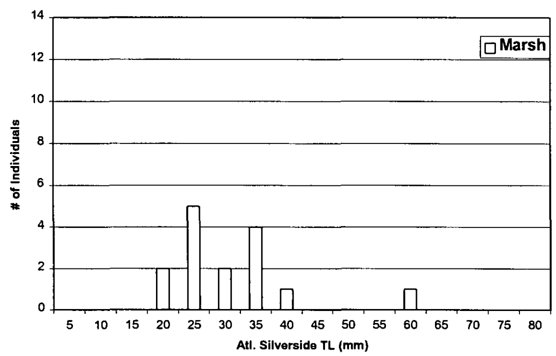
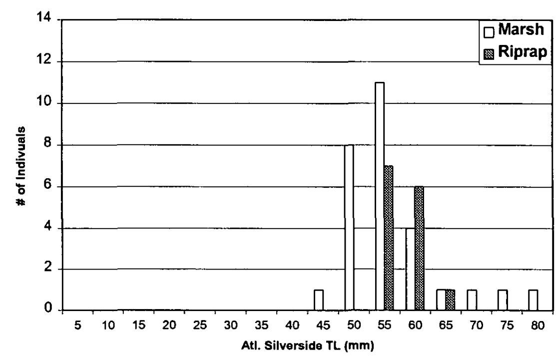


Aug 27 - Sep 2



LITERATURE CITED

Able, K.W. 1999. Measures of juvenile fish habitat quality: Examples from a National Estuary Research Reserve. In: L. R. Benaka ed. Fish habitat: Essential habitat and rehabilitation, p. 134-147. American Fisheries Society. Bethesda, Maryland.

Able, K.W., J.P. Manderson and A.L.Studholme. 1998. The distribution of shallow water juvenile fishes in an urban estuary: The effects of manmade structures in the lower Hudson River. Estuaries. 21: 731-744.

Able, K.W. and S.M. Hagan. 2000. Effects of common reed (*Phragmites australis*) invasion on marsh surface macrofauna: Response of fishes and decapod crustaceans. Estuaries. 23: 633-646.

Abraham, B.J. 1985. Species profiles: Life histories and environmental requirements of coastal fishes and invertebrates (Mid-Atlantic). Mummichog and Striped Killifish. U.S. Fish and Wildlife Service Biological Report FWS/OBS-82/11.40. 23p.

American Society of Ichthyologists and Herpetologists, American Fisheries Society, American Institute of Fisheries Research Biologists. 1988. Guidelines for use of fishes in field research. Fisheries. 13: 16-23.

Anderson, G. 1985. Species profiles: Life histories and environmental requirements of coastal fishes and invertebrates (Gulf of Mexico). Grass Shrimp. U.S. Fish and Wildlife Service Biological Report FWS/OBS-82/11.35. 17p.

Austin, H.M., D.M Seaver, and C.M. Wagner. 1996. Monitoring juvenile recreational fishes on the ocean beaches and Eastern Shore of Virginia with special focus on Developing a Bluefish, *Pomatomus saltatrix*, young-of-the-year index in Virginia. Virginia Marine Resources Commission, contract no. RF-96-09. 115p.

Ayers, L.A. 1995. Finfish communities of two intertidal marshes of the Goodwin Islands, York River, Virginia. M.A. Thesis, The College of William and Mary, Williamsburg, VA. 62p.

Baltz, D.M., C. Rakocinski, and J.W. Fleeger. 1993. Microhabitat use by marsh-edge fishes in a Louisiana estuary. Environmental Biology of Fishes. 36: 109-126.

Barnard, J. 1994. New brochure encourages "green" solutions to shoreline erosion. Virginia Institute of Marine Science, Virginia Wetlands Report. Fall, No. 94-9.

Barnard, T.A. Jr., P. Clancy, K. Duhring, K.J. Havens, C. Hershner, P. Mason, D. O'Brien, W.I. Priest III, W. L. Roberts, L.M. Varnell, and D. Weiss. 2001. VIMS shoreline permit database impacts and alterations summary 1993-2000. Virginia Institute of Marine Science, Wetlands Program. Technical Report 01-4.

Beauchamp, D. A., E.R. Byron, and W.A. Wurtsbaugh. 1994. Summer habitat use by littoral zone fishes in Lake Tahoe and the effects of shoreline structures. North American Journal of Fisheries Management. 14: 385-394.

Beck, M.W., K.L. Heck, K.W. Able, D.L. Childers, D.B. Eggleston, B.M. Gillanders, B. Halpern, C.G. Hays, K. Hoshino, T.J. Minello, R.J. Orth, P.F. Sheridan and M.P. Weinstein. 2001. The identification, conservation and management of estuarine and marine nurseries for fish and invertebrates. Bioscience. 8: 633-641.

Berman, M., H. Berquist and C. Hershner. 1998. Quantitative evaluation of the regulatory process and its impacts to tidal wetlands. Comprehensive Coastal Inventory, Center for Coastal Management and Policy, Virginia Institute of Marine Science, Gloucester Point, Virginia. Final report to US EPA Region III, Philadelphia, PA. Grant No. CD993277-01-0.

Berman, M, H. Berquist, S. Dewing, G. Thomas and R. Laird. 1999. Gloucester county shoreline and tidal marsh inventory. Comprehensive Coastal Inventory, Center for Coastal Management and Policy, Virginia Institute of Marine Science, Gloucester Point, Virginia. Special Report in Applied Marine Science and Ocean Engineering # 351.

Boesch, D.F., and R.E. Turner. 1984. Dependence of fishery species on salt marshes: the role of food and refuge. Estuaries. 7: 460-468.

Byrne, D.M. 1995. The effect of bulkheads on fish distribution and abundance: A comparison of littoral fish and invertebrate assemblages at bulkheaded and non-bulkheaded shorelines on a Barnegat Bay lagoon. Proceedings of the Second Marine and Estuarine Shallow Water Science and Management Conference, p. 53-56. April 3-7, Atlantic City, New Jersey. United States EPA.

CBNERRVA. 2001. York River juvenile summer flounder survey. Chesapeake Bay National Estuary Research Reserve in Virginia.

Cicchetti, G. 1998a. Habitat use, secondary production, and trophic export by salt marsh nekton in shallow waters. Doctoral Dissertation, The College of William and Mary, Williamsburg, VA. 276p.

Cicchetti, G. 1998b. The importance of edge in creation and management of salt marshes for commercially valuable marine transient species. Proceedings of the 16th International Conference of the Coastal Society, p 353-358. Williamsburg, VA.

Cicchetti, G. 1998c. Use of salt marsh edge by economically valuable fishes and crustaceans. Final Research Report to the Office of Ocean and Coastal Resource Management. NOAA/SRD Award # 97-066-CBV. Silver Spring, MD. 28p.

Cicchetti, G. and R.J. Diaz. 2000. Types of salt marsh edge and export of trophic energy from marshes to deeper habitats. In M.P. Wienstein and D.A. Kreeger, eds. Concepts and Controversies in Tidal Marsh Ecology, p. 515-541. Kluwer Academic Publishers, Boston.

Clarke, K.R. and R. M. Warwick. 1994. Change in Marine Communities: An Approach to Statistical Analysis and Interpretation. Plymouth Marine Laboratory, Plymouth, U.K.

Conover, D.O. and M.R. Ross. 1982. Patterns in seasonal abundance, growth and biomass of the Atlantic silverside, *Menidia menidia*, in a New England estuary. Estuaries. 7: 275-286.

Craig, K.J and L.B. Crowder. 2000. Factors influencing habitat selection in fishes with a review of marsh ecosystems. In M.P. Wienstein and D.A. Kreeger, eds. Concepts and Controversies in Tidal Marsh Ecology, p. 241-261. Kluwer Academic Publishers, Boston.

Deegan, L.A. 1989. Nekton, the free-swimming consumers. In J.W. Day, C.S. Hall, W.M. Kemp and A.Y. Arancibia, eds. Estuarine Ecology, p.377-437. John Wiley and Sons, Inc.

Dittel, A.I., A.H. Hines, G.M. Ruiz, and K.K. Ruffin. 1995. Effects of shallow water refuge on behavior and density-dependent mortality of juvenile blue crabs in Chesapeake Bay. Bulletin of Marine Science. 57: 902-916.

Fitz, H.C. and R.C. Wiegert. 1991. Utilization of the intertidal zone of a salt marsh by the blue crab *Callinectes sapidus*: density, return frequency, and feeding habits. Marine Ecology Progress Series. 76: 249-260.

Folk, R.L. 1980. Petrology of sedimentary rocks. Hemphill Pub. Co. Columbus, Ohio.

Hales, L.S. and M.J. Van Den Avyle. 1989. Species profiles: Life histories and environmental requirements of coastal fishes and invertebrates (South Atlantic). Spot. U.S. Fish and Wildlife Service Biological Report FWS/OBS-82/11.91. 24p.

Halpin, P.T. 2000. Habitat use by an intertidal salt-marsh fish: trade-offs between predation and growth. Marine Ecology Progress Series. 198: 203-214.

Hardaway, C.S. and R.J. Byrne. 1999. Shoreline management in Chesapeake Bay. Special Report in Applied Marine Science and Ocean Engineering No. 356, Virginia Institute of Marine Science, Gloucester Point, VA. 54p.

Hardy, J.D. 1978. Development of fishes of the Mid-Atlantic Bight: an atlas of egg, larval and juvenile stages, v. II, Anguillidae through Syngnathidae. U.S. Fish and Wildlife Service Biological Services Program. FWS/OBS-78/12. 445p.

Hay, M.E. and J.P. Sutherland. 1988. The ecology of rubble structures of the South Atlantic Bight: A community profile. U.S. Fish and Wildlife Service Biological Report FWS/OBS-85/7.20. 67p.

Hendon, J.R., M.S. Peterson, and B.H. Comyns. 2000. Spatio-temporal distribution of larval *Gobiosoma bosc* in waters adjacent to natural and altered marsh-edge habitats of Mississippi coastal waters. Bulletin of Marine Science. 66: 143–156.

Hettler, W.J. Jr. 1989. Nekton use of regularly-flooded saltmarsh cordgrass habitat in North Carolina, USA. Marine Ecology Progress Series. 56: 111-118.

Herke, W.H., E.E. Knudsen, P.A. Knudsen, and B.D. Rogers. 1992. Effects of semi-impoundment of Louisiana marsh on fish and crustacean nursery use and export. North American Journal of Fisheries Management. 12: 151-160.

Hines, A.H. and G.M. Ruiz. 1995. Temporal variation in juvenile blue crab mortality: nearshore shallows and cannibalism in Chesapeake Bay. Bulletin of Marine Science. 57: 884-901.

Holmquist, R.K. 2001. Life history attributes of Mid-Atlantic *Menidia menidia* (Pisces: Atherinidae) and a comparison with (Northern Massachusetts) and Southern (South Carolina) populations. M.S. Thesis, The College of William and Mary, Williamsburg, VA. 74p.

Howe, J.C., R.K. Wallace, and R.S. Rikard. 1999. Habitat utilization by postlarval and juvenile penaeid shrimps in Mobile Bay, Alabama. Estuaries. 22: 971-979.

Jennings, M.J., M.A. Bozek, G.R. Hatzenbeler, E.E. Emmons and M. D. Staggs. 1999. Cumulative effects of incremental shoreline habitat modification on fish assemblages in north temperate lakes. North American Journal of Fisheries Management. 19: 18-27.

Kneib, R.T. 1982. The effects of wading birds (*Ardeidaie*) and blue crabs (*Callinectes sapidus*) on the population size and structure of the common mummichog (*Fundulus heteroclitus*). Estuarine, Coastal and Shelf Science. 14: 159-165.

Kneib, R.T. 1987. Predation risk and use of intertidal habitats by young fishes and shrimp. Ecology. 68: 379-386.

Kneib, R.T. 1997. The role of tidal marshes in the ecology of estuarine nekton. In A.D. Ansell, R.S. Gibson and M. Barnes, eds. Oceanography and Marine Biology, An Annual Review. 35, p 163-220. Argyll, Scotland.

Kneib, R.T. 2000. Salt marsh ecoscapes and production transfers by estuarine nekton in the southeastern United States. In Weinstein, M.P. and D.A. Kreeger, eds. Concepts and Controversies in Tidal Marsh Ecology, p. 267-290. Kluwer Academic Publishers, Boston.

Kneib, R.T., and A.E. Stiven. 1982. Benthic invertebrate response to size and density manipulations of the common mummichog, *Fundulis heteroclitus*, in an intertidal salt marsh. Ecology. 63: 1518-1532.

Kneib, R.T. and S.L. Wagner. 1994. Nekton use of vegetated marsh habitats at different stages of tidal inundation. Marine Ecology Progress Series. 106: 227-238.

Lotrich, V.A. and W.H. Meredith. 1974. A technique and the effectiveness of various acrylic colors for subcutaneous marking of fish. Transactions of the American Fisheries Society. 1: 140-142.

McIvor, C.C., and W.E. Odum. 1986. The flume net: A quantitative method for sampling fishes and macrocrustaceans on tidal marsh surfaces. Estuaries. 9: 219-224.

McIvor, C.C., and W.E. Odum. 1988. Food, predation risk, and microhabitat selection in a marsh fish assemblage. Ecology. 69: 1341-1351.

McIvor, C.C. and L.P. Rozas. 1996. Direct nekton use of intertidal saltmarsh habitat and linkage with adjacent habitats: A review from the southeastern United States. In K.F. Nordstrom and C.T. Roman, eds. Estuarine Shores: Evolution, Environments and Human Alterations, p. 311-344. John Wiley and Sons, New York.

Middleton, R.W. 1986. The seasonal and diel use by juvenils and adult finfish of a mesohaline intertidal creek on the York River, Virginia. Doctoral Dissertation, The College of William and Mary. 104p.

Miltner, R.J., S.W. Ross and M.H. Posey. 1994. Influence of food and predation on the depth distribution of juvenile spot (*Leiostomus xanthurus*) in tidal nurseries. Canadian Journal of Fisheries and Aquatic Science. 52. 971-982.

Minello, T.J., R.J. Zimmerman, and R. Medina. 1994. The importance of edge for natant macrofauna in a created salt marsh. Wetlands 14: 184-198.

Minello, T.J. and J.W. Webb, Jr. 1997. Use of natural and created *Spartina alterniflora* salt marshes by fishery species and other aquatic fauna in Galveston Bay, Texas, USA. Marine Ecology Progress Series. 151: 165-179.

Minton, M.D. 1999. Coastal wetland restoration and its potential impact on fishery resources in the Northeastern United States. In L. R. Benaka, ed. Fish habitat: Essential habitat and rehabilitation, p. 405-420. American Fisheries Society. Bethesda, Maryland.

Mulvihill, E.L., C.A. Francisco, J.B. Glad, K.B. Kaster, and R.E. Wilson. 1980. Biological impacts of minor shoreline structures on the coastal environment: State of the art review. U.S. Fish and Wildlife Service, Biological Services Program. FWS/OBS-77/51. 153p.

Murdy, E.O., R.S. Birdsong, and J.A. Musick. 1997. Fishes of Chesapeake Bay. Smithsonian Institution. 324p.

Nordstrom K.F. and C.T. Roman. 1996. Emerging research needs related to interactions on estuarine shores. In K.F. Nordstrom and C.T. Roman, eds. Estuarine Shores: Evolution, Environments and Human Alterations, p. 468-478. John Wiley and Sons, New York.

Olmi, E.J. and J.M. Bishop. 1983. Variations in total width-weight relationships of blue crabs, *Callinectes sapidus*, in relation to sex, maturity, molt stage, and carapace form. Journal of Crustacean Biology. 3:575-581.

Orson, R., W. Panageotou, and S.P. Leatherman. 1985. Response of tidal salt marshes of the U.S. Atlantic and Gulf coasts to rising sea levels. Journal of Coastal Research. 1: 29-37.

Peterson, G.W., and R.E. Turner. 1994. The value of salt marsh edge vs. interior as a habitat for fish and decapod crustaceans in a Louisiana tidal marsh. Estuaries. 17: 235-262.

Peterson, M.S., B.H. Comyns, J.R. Hendon, P.J. Bond, and G.A. Duff. 2000. Habitat use by early life-history stages of fishes and crustaceans along a changing estuarine landscape: Differences between natural and altered shoreline sites. Wetlands Ecology and Management. 8: 209-219.

Pielou, E.C. 1975. Ecological Diversity. John Wiley & Sons, Inc., New York. 165p.

Priest, W.I., K.J. Havens, T.A. Barnard, J.G. Bradshaw, and M. Wohlgemuth. 1990. Cumulative impacts of shoreline construction activity on tidal wetlands in Virginia. Virginia Institute of Marine Science, Wetlands Program. Technical Report No. 90-3, August. 8p.

Rakocinski, C.F., D.M. Baltz, and J.W. Fleeger. 1992. Correspondence between environmental gradients and the community structure of marsh-edge fishes in a Louisiana estuary. Marine Ecology Progress Series. 80: 135-148.

Roman, C.T. and N. F. Nordstrom. 1996. Environments, processes and interactions of estuarine shores. In K.F. Nordstrom and C.T. Roman, eds. Estuarine Shores: Evolution, Environments and Human Alterations, p. 1-13. John Wiley and Sons, New York.

Rozas, L.A. 1992. Comparison of nekton habitats associated with pipeline canals and natural channels in Louisiana salt marshes. Wetlands 12: 136-146.

Rozas, L.A. 1995. Hydroperiod and its influence on nekton use of the salt marsh: a pulsing ecosystem. Estuaries. 18: 579-590.

Rozas, L.P. and W.E. Odum. 1987. The role of aquatic vegetation in influencing the abundance of nekton on contiguous tidal fresh-water marshes. Journal of Experimental Marine Biology and Ecology. 114: 289-300.

Rozas, L.P., C.C. McIvor and W.E. Odum. 1988. Intertidal rivulets and creekbanks: corridors between tidal creeks and marshes. Marine Ecology Progress Series. 47: 303-307.

Rozas, L.A., and D.J. Reed. 1993. Nekton use of marsh-surface habitats in Louisiana (USA) deltaic salt marshes undergoing submergence. Marine Ecology Progress Series. 96: 147-157.

Rozas, L.A., and T.J. Minello. 1997. Estimating densities of small fishes and decapod crustaceans in shallow estuarine habitats: a review of sampling design with focus on gear selection. Estuaries. 20: 199-213.

Ruiz, G.M, A.H. Hines and M.H. Posey. 1993. Shallow water as a refuge habitat for fish and crustaceans in non-vegetated estuaries: an example from Chesapeake Bay. Marine Ecology Progress Series 99:1-16.

Sarthou, C.M. 1999. An environmentalist's perspective on essential fish habitat. In L. R. Benaka, ed. Fish Habitat: Essential Habitat and Rehabilitation, p. 11-23. American Fisheries Society, Bethesda, Maryland.

Schmetterling, D.A., C.G. Clancy, and T.M. Brandt. 2001. Effects of riprap bank reinforcement on stream salmonids in the western United States. American Fisheries Society. 26: 6 – 11.

Smith, S.M., J.G. Hoff, S.P. O'Neil and M.P. Weinstein. 1984. Community and trophic organization of nekton utilizing shallow marsh habitats, York River. Fishery Bulletin 82: 455- 466.

State of the York Watershed 2000. The York Watershed Council. Comprehensive Coastal Inventory, Virginia Institute of Marine Science, Gloucester Point, VA.

Stevenson, J.C. and M.S. Kearney. 1996. Shoreline dynamics on the windward and leeward shores of a large temperate estuary. In K.F. Nordstrom and C.T. Roman, eds. Estuarine shores: Evolution, Environments and Human Alterations, p. 233 - 259. John Wiley and Sons, New York.

Thayer, G.W., W.E. Schaaf, J.W. Angelovic and M.W. LaCroix. 1973. Caloric measurements of some estuarine organisms. Fishery Bulletin. 71: 289-296.

Titus, J. G. 1998. Rising seas, coastal erosion, and the takings clause: How to save wetlands and beaches without hurting property owners. Maryland Law Review. 57:1279-1399.

VMRC. 1999. Shoreline development BMP's. Habitat Management Division, Virginia Marine Resources Commision, August 1999.

Varnell, L.M. and K.J. Havens. 1995. A comparison of dimension-adjusted catch data methods for assessment of fish and crab abundance in intertidal salt marshes. Estuaries. 18: 319-325.

Weisberg, S.B. and V.A. Lotrich. 1982. The importance of an infrequently flooded intertidal marsh surface as an energy source for the mummichog *Fundulus heteroclitus*: An experimental approach. Marine Biology 66: 307-310.

Welsh, B.L. 1975. The role of grass shrimp, *Palaemonetes pugio,* in a tidal marsh ecosystem. Ecology. 56: 513-530.

West, J.M. and J.B. Zedler. 2000. Marsh-creek connectivity: Fish use of a tidal salt marsh in Southern California. Estuaries. 23: 699-710.

Wohlegemuth, M. 1990. Tidal wetland values. Virginia Institute of Marine Science, Wetlands Program. Technical Report No. 90-5. October. 8p.

Yozzo, D.J., A. Mannini, and D.E. Smith. 1994. Mid-summer abundance of resident sub adult marsh nekton at the Virginia Coast Reserve. Virginia Journal of Science. 45: 21-29.

Yozzo, D.J. and D.E. Smith. 1998. Composition and abundance of resident marsh-surface nekton: Comparison between tidal freshwater and salt marshes in Virginia, USA. Hydrobiologia. 362: 9-19.

Zimmerman, R.J., and T.J. Minello. 1984. Densities of *Penaeus aztecus, Penaeus setiferus*, and other natant macrofauna in a Texas salt marsh. Estuaries 7: 421-433.

Zimmerman, R.J., T.J. Minello and L.P. Rozas. 2000. Salt marsh linkages to productivity of penaeid shrimps and blue crabs in the northern Gulf of Mexico. In Weinstein, M.P. and D.A. Kreeger, eds. Concepts and Controversies in Tidal Marsh Ecology, p.293-310. Kluwer Academic Publishers, Boston.

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